Habitat fragmentation and its effects on birds and grasshoppers in eucalypt remnants in the Tasmanian Midlands.

by

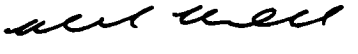
Andrew
Michael MacDonald, BSc (hons)

Submitted in fulfilment of the requirements for the degree of Doctor of Philosophy

School of Geography and Environmental Studies
Faculty of Science and Engineering
University of Tasmania

August, 2001

I declare that this thesis contains no material which has been accepted for a degree or diploma at the University or any other institution, except by way of background information and duly acknowledged in the thesis, and to the best of my knowledge and belief no material previously published or written by another person except where due acknowledgment is made in the text of the thesis.

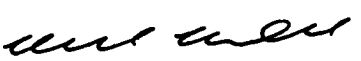


Michael MacDonald

Date: 2/8/01

Authority of Access

This thesis may be made available for loan and limited copying in accordance with the Copyright Act 1968



Michael MacDonald

Date: 2/8/01

Abstract

In the subhumid Midlands region of Tasmania, forty-seven remnants of eucalypt woodland and dry sclerophyll forest as well as six control sites were surveyed for birds and grasshoppers, in order to examine the responses of these groups to habitat fragmentation. A wide range of characteristics of the remnants (including position in landscape, management regime and vegetation) were related to species richness, diversity and density of the two groups as well as to the distributions of individual species. Remnant size, vegetation structure and tree health all showed significant relationships with bird species richness and diversity. Small remnants with open understorey and high levels of dieback showed radically different species composition than larger ones or those with dense understorey. Where the understorey is open the noisy miner (*Manorina melanocephala*) is present in colonies and is able to effectively exclude almost all other species by concerted aggressive behaviour. Noisy miner colonies were associated with small remnants but were also present at the edges of larger remnants, where proximity to open country and vegetation structure both predicted their presence. Interspecific competition is considered to be the major determinant of species richness and of many species' distributions in the study area. Analysis of remnants which are not dominated by noisy miner colonies found that area and isolation were significantly correlated with species richness and diversity, although larger remnants did not have more species at a given point. Summer migrants and nomadic species are considered to be more sensitive to habitat fragmentation as a result of the presence of noisy miner colonies in the fragmented habitat. Interspecific competition is considered to be the driving force behind avifaunal trends in the study area. Grasshopper species richness was not related to any of the variables measured, but diversity was higher in remnants in better condition, while density was higher in remnants in poor condition. Common grasshopper species responded to a range of variables. Management of remnants for conservation of avifaunal and grasshopper values is discussed. In the context of avifauna conservation, it is suggested that although larger remnants are more likely to support a healthy suite of bird species, the presence of a dense understorey in smaller remnants can improve their conservation value. Both of these options are likely to also lead to the maintenance of forest- and woodland-dwelling grasshopper species.

Acknowledgments

I would like to thank my supervisor, Jamie Kirkpatrick, for his patience, assistance and encouragement in this research.

Peter McQuillan identified grasshoppers and provided considerable assistance in understanding the results of the grasshopper collection.

All landowners involved have been helpful in allowing me access to their properties.

Other postgraduate students have provided advice and help at times. In particular, Mona Loofs has helped me to maintain my sanity.

I wish to thank my family for their support over the past five years.

Anneli MacDonald has inspired me and enabled me to finish this and keep my promises.

Contents (cont.)

Figures

Tables

Contents (cont.)

1. A review of the effects of habitat fragmentation

1.1 Introduction

A worldwide threat to natural habitats and ecosystem functioning is fragmentation. This seems like a self-evident statement, but when one looks at it a little more closely it is not quite so obvious. *The new shorter Oxford dictionary* (1993) defined fragmentation in a biological sense as "separation into parts which form new individuals or units." This has been equated in many ecological studies to the subdivision of continuous habitat into a series of smaller components (often referred to as habitat islands) at a landscape level. However, Lord & Norton (1990) preferred to define fragmentation as the disruption of ecosystem continuity, which is not limited to any scale as ecosystems function across a wide range of scales. Individual plants may be considered as habitat fragments at large scales, but to a large extent the concept of fragmentation has been coloured by its importance in conservation issues. This has arisen through the widespread clearance of natural vegetation and the conversion of land to other purposes, including urban areas, mining, and, most extensively, agriculture. This process has transformed large areas of contiguous native vegetation into isolated remnants in a matrix of cultural vegetation, subject to different influences and stresses, as discussed below.

It is worth noting that very few (if any) natural systems are completely homogeneous and completely contiguous, and that for some fragmentation is an integral part of their being. Alpine habitats are often naturally fragmented because the conditions required for their presence are unevenly distributed. Fire can play a major role in natural fragmentation, as with the rainforest-buttongrass mosaic of south-western Tasmania. Janzen (1983) presented tree falls in tropical forest in Costa Rica as an example of fragmentation within a habitat. However, when ecosystems are naturally fragmented we tend to refer to them as mosaics, a small example of the emotive power of labels. The fact that fragmentation can be considered a natural state in many instances is not to say that one should dismiss the negative effects of human-induced fragmentation. It is merely to state that fragmentation, as a concept, is not simple.

Having put forward fragmentation as a natural state, it should be emphasised that the damage to natural systems caused by human activity is undeniable and considerable and

deserving of attention. Human activities have caused fragmentation of natural habitats for thousands of years. With increased population growth the rate has increased markedly. The greatest cause of fragmentation has been land clearance for agricultural purposes, but other means have been the use of fire, logging activities, and urban spread. These activities are unlikely to decrease in the foreseeable future. Fragmentation of previously contiguous habitats has emerged as one of the major conservation problems of the present time, and for this reason it has become a focus of many studies of ecosystems.

It is important to emphasise that the impacts of fragmentation do not necessarily stem simply from absolute loss of habitat. Habitat loss is a problem in itself, but frequently there are more severe or more complex (or both) results than would be expected from habitat loss alone. Saunders *et al.* (1991) reviewed the effects of habitat fragmentation on forests and concluded that changes in microclimate, influence of external factors, and degree of isolation are the major results. The effects on microclimate include changes to radiation/evapotranspiration levels, to wind profiles and to hydrological cycles. The isolation of a fragment is not just a function of distance from similar habitat, but is also influenced by the nature of the intervening habitat. External influences may include increased predation and invasion of species. Fragment shape can also be important, as it affects the proportion of that fragment which is subject to edge effects. Once these factors start to affect the present biota, the flow-on effects can be significant. For example, Diamond *et al.* (1987) ascribed the disappearance of a parasitic bird species in an isolated forest on Java to the loss of host species which were unable to survive in that remnant.

The scale of fragmentation is important, as different species perceive and respond to the same environment at different scales. A wide ranging raptor may utilise a range of discontinuous habitats (from the human perspective) without difficulty, while a small passerine species reluctant to cross open country will have a different response to exactly the same landscape. In Quebec, Desrochers & Hannon (1997) found that songbirds were generally more reluctant to cross a 50 m gap than 50 m of forest, but that species differed greatly in their responses to gaps. Margules *et al.* (1994) found different responses from two small invertebrates in forest fragments as a result of differences in their ecological and biological characteristics, and in their evolutionary

ages. Humans tend to look at fragmentation on a landscape level, because this seems to be the level at which we perceive it, but from the point of view of the organisms concerned, this may be too narrow or too broad. Lord & Norton (1990) discussed this issue thoroughly.

The response of an ecosystem to fragmentation can occur at many levels. The study of plant or animal community responses, which has proven useful in other areas, has been generally found to be inappropriate to measure the effects of fragmentation. This is because the responses to fragmentation have been mostly found to be at the species or population level and community-level responses have been difficult to find. Because taxa respond variably to fragmentation, certain elements of an existing ecosystem will respond differently to the same ecological change. Fragmentation may not be detrimental to all elements of an ecosystem; rather, it can alter the balance within the ecosystem by favouring certain elements over others.

The importance of local factors has been somewhat overlooked in the quest for a universal theory regarding habitat fragmentation. However, as a result of the primacy of species-level responses, forest fragmentation is more likely to have different effects from continent to continent or from region to region. For example, Howe *et al.* (1981) found that rainforest bird species in New South Wales are good dispersers, probably because of the historically discontinuous distribution of rainforest on the Australian continent. As a result, even very small remnants of rainforest contained some species typical of larger rainforest areas. Schieck *et al.* (1995) found that old-growth bird species richness and abundance in montane forest patches on Vancouver Island were not related to patch size, and that this may have been due to the fact that they had evolved in heterogeneous forests and therefore were less susceptible to fragmentation. Alternatively, Telleria & Santos (1995) found that the bird species in small forests in Spain were a nested subset of those found in larger areas. Local circumstances can also include the nature of the intervening habitat. Forest fragments surrounded by completely cleared land will function differently from those between which the cleared land also includes elements of suitable habitat, for example, hedgerows or isolated trees (McIntyre & Barrett 1992).

As an optimistic aside on this topic, most studies have assumed the attitude that fragmentation is irreversible, which is simply not the case. The concept of habitat corridors which may (or may not, depending on your viewpoint) link fragments and increase contiguous habitat aside, some approximation of natural habitat can be restored whether intentionally or not. The area of Harvard Forest, Massachusetts, was 90% forested in 1771. Following European settlement this fell to 10% by 1850, but abandonment of farms meant that by 1956 it was 85% forested (Burgess & Sharpe 1981). Whether this was functionally the same forest as previously is questionable, but the concept of habitat loss as unidirectional is not correct.

The history of research into the effects of habitat fragmentation has to a large extent been the history of research into bird populations and communities in fragments of forest caused by land clearance in agricultural regions. This is both a strength and a failing. It is a strength because it has provided an extensive body of knowledge about one taxonomic group, and this has allowed comparisons to be made over time and space and allowed a sophisticated debate to arise. It is a failing, however, because the responses of birds to fragmentation have sometimes been accepted as representative of all taxonomic groups in all situations, and this has narrowed our understanding of fragmentation to some extent. Intuition tells us that it is very unlikely that one taxon could represent all others, and the representativeness of birds has been questioned by a number of studies involving other taxa (Margules *et al.* 1994, Robinson *et al.* 1992, Wilcox *et al.* 1986). Even within avifauna the situation is complicated and variable both temporally and spatially as different species react differently to the effects of fragmentation. "Appraisal of total biodiversity cannot rely on an examination of single taxa such as birds because biodiversity relies on interactions between taxa and ecosystem structures, and their history" (Hansson 1997).

The reasons for the historical emphasis on birds are several. Birds, being a mobile and eye-catching element of the biota, are relatively easy to observe, and their habitat requirements are relatively well known. The effects of fragmentation on birds have been qualitatively noticeable in some areas before studies have been performed. There is also a certain degree of self-perpetuation, because the large body of work already completed on birds can stimulate further work. As a group birds are also interesting because many species are area-demanding, while also being very mobile and therefore able to meet

their area demands from a range of sites (Rolstad 1991). I would also argue that the scale factor is important, as birds seem to be affected by fragmentation on a level that is easily comprehensible to humans. This may be a result of the relatively large body size of birds and their position high up in the food chain.

Studies of other taxa have considerable value because "goals catering to insect or plant conservation may thus conflict with strategies designed solely for conservation of birds or mammals" (Whitcomb 1987). This is because they may have different life history strategies. Ideally, these need to be understood for all animals and plants, and realistically they should be understood for as wide a range of groups as possible. In addition, these studies can illuminate other trends which may not be present within birds. Research into the effects of habitat fragmentation on various taxa is increasing. There is still a bias towards vertebrates, largely as a result of better general understanding and ease of sampling of this group. Patch occupancy by the Columbian ground squirrel (*Spermophilus columbianus*) in northern USA was related to distance from a source of squirrels, but not to area, partly because squirrels tended to settle near other squirrels rather than in vacant patches (Weddell 1991). Verboom & Van Apeldoorn (1990) found that area and isolation of patches affected presence and absence of red squirrels (Sciurus vulgaris). Deacon & Mac Nally (1998) looked at mammals in fragments if eucalypt forest in Victoria and concluded that the fauna of smaller fragments was a nested subset of larger ones as a result of local extinction, which in turn is a product of the biological characteristics of various species which make them vulnerable to extinction. In the Western Australian wheatbelt Kitchener (1982) found that the presence of bird and mammal species were best explained by area, while plant associations were more important for lizards.

Robinson *et al.* (1992) experimentally created fragments of grasslands, and concluded that different taxa responded differently to the fragmentation. Mammals and snakes were very sensitive to patch size, while the results for arthropods were equivocal. They warned of the use of species diversity as a measure of habitat quality, preferring rather to compare the population changes of individual species. Characteristics of montane butterfly species (particularly vagility) in the Great Basin in North America were important in determining their response to habitat area, while neither distance to other montane areas nor habitat diversity showed significant relationships (Wilcox *et al.* 1986).

Trapdoor spiders in habitat fragments in Western persist well in small patches, probably due to long life cycle and sedentary habits (Main 1987). Abensperg-Traun *et al.* (1996) considered that disturbance played a greater role than position in landscape in determining species richness of various arthropod groups, while disturbance indices (weed cover, sheep pellets) were significantly higher in small and poorly-connected remnants. These findings were supported by Scougall *et al.* (1993) in the Western Australian wheatbelt, who found fencing of remnants lead to better habitat condition and lower levels of the aggressive dominant ant functional group *Iridomyrmex* spp. This wide variety of results demonstrates the value of examining trends from a range of animal groups, as these may experience fragmentation on a different scale, or may respond to it differently, both as species and at higher groupings. This is likely because of the wider array of life history characteristics present.

1.2 Island biogeography and other theories

A significant reason for the interest in birds on fragments of natural vegetation has been the existence of the theory of island biogeography (MacArthur & Wilson 1967). This was developed as a result of work on birds of archipelagoes in an attempt to more fully explain concepts introduced by Arrhenius (1921) and Gleason (1922) regarding the tendency of larger areas to hold more species. Broadly speaking the theory predicts that the biota of an island will reach an equilibrium number of species over time, with this number depending on the size of the island and its degree of isolation, and with immigration and extinction the driving forces. Debate over the merits of island biogeographic theory has been quite heated in scientific terms (e.g. Simberloff & Abele 1982, Gilbert 1980) and it is certainly by no means universally accepted. Different reviews (Andren 1994, Rolstad 1991) have found that support for island biogeography theory varies between studies, although the tendency has been towards rejection of the theory.

Island biogeographic theory has been largely discredited as applying to habitat patches. Criticism of the application of this theory to habitat patches on continents has largely concentrated on the fact that they are not analogous to true islands. The idea of fragmentation producing island analogues implies that these patches of habitat are

separated from each other by habitat that is as hostile and inhospitable to the organisms using the patches as the ocean is to land birds on islands, and this is clearly not the case (McCoy 1982, McIntyre & Barrett 1992). It is also not the case for most other animals. Even if it may be applicable for some bird species (and in the northern NSW tablelands, McIntyre & Barrett (1992) concluded that only around 10% of bird species were truly living in a fragmented habitat) it is not the case for all, and this discredits the idea of predicting species numbers from the theory. Janzen (1983) said that "the conventional island is not surrounded by a habitat rich in organisms competent to forage extensively on the island...and prone to bombard the island with juveniles quite capable of taking up residence." In the case of birds the evidence is that a significant proportion of species are either able to or prefer to use a range of habitats within the total landscape.

In particular, island biogeographic theory is not applicable to long distance migrant birds, which recolonise areas of forest every year, and as such cannot be considered subject to the concepts of isolation (and the use of single habitat patches by each individual) as applying to island biogeographic theory (Ambuel & Temple 1983). Metapopulation theory has also questioned island biogeographic theory, in the context of local and regional immigration and extinction, by presenting evidence that local extinction of species is a natural state of affairs in some cases. In North America, Donovan *et al.* (1997) described small forest fragments as population sinks for migratory birds, while contiguous forest acts as a population source, on the basis of nesting success, although Friesen *et al.* (1999) disputed that this was the case for all migrants. Certain species would most likely become extinct in fragments without immigration, although there is the possibility of their populations stabilising at lower levels.

The idea that small patches of forest are (or will be) unable to sustain populations of given species has been popular, but seems likely to be valid only for that small number of species which actually experience demographic isolation as a result of fragmentation or which are reduced below critical population levels as a result of habitat loss. For other species, regional population dynamics may be more important than local, and many species are able to use a range of habitat patches, as well as the intervening land. Saunders *et al.* (1991) said that "presence of a species in a remnant is no guarantee of its continued existence: successful reproduction is required." But nor is local extinction irreversible. In Victoria, Bennett and Ford (1997) considered that "single fragments in

disturbed landscapes, such as patches of forest or woodland in an agricultural mosaic, are rarely large enough to support populations that are self-sustaining in the long term." However, they recognised that this does not necessarily mean that these patches are of no use to either individuals or populations of a species, Most bird species interact with the surrounding landscape to varying degrees, and different remnants may be used for different resources by the same populations or individuals. Lynch (1987) put forward this idea in the North American context, suggesting that few forest patches can maintain a stable avifauna by themselves, but that even the smallest patches can be important for exchange of individuals and genetic material. The fact that not every species is present in a given patch every year does not diminish its value in this process. Modelling by Howe et al. (1991) who concluded that "although these demographic sinks are unable to persist independently, they may contribute significantly to metapopulation size and longevity" also supported the metapopulation concept. From a conservation point of view, the danger in this is that sites which may seem to be self-sustaining may be depending on influxes from elsewhere, so local populations may disappear as a result of habitat loss elsewhere (Temple & Carey 1988). The landscape spatial structure is important for metapopulations (Fahrig & Merriam 1994). If the structure restricts movement between patches then the area required for population survival is large.

The importance of recolonisation has been demonstrated by Diamond *et al.* (1987) who studied the avifauna of an 86 ha forest in Java which had been isolated for fifty years. The two major variables explaining persistence of species were initial population size and abundance in the surrounding countryside. "In short, small population size is a good predictor of extinction for populations that are effectively isolated...but not for populations maintained by a nearby source of colonists." Stacey & Taper (1992) looked at the acorn woodpecker (*Melanerpes formicivorus*), which is a species generally found in small isolated populations but which is not rare or endangered. When they modelled persistence of a population they obtained results of 16-49 years, until they included immigration in the model, when the results were 1000+ years, even when immigration was at low levels. Quinn & Hastings (1987) examined six empirical studies, four on birds, one on vascular plants and one on zooplankton and found that there is no uniform tendency for populations to persist for longer or shorter times as a result of habitat subdivision. This was also supported by Higgs (1981). Quinn & Hastings (1987) suggested that because of the rapid change of habitats, whether natural or not, 100 years

persistence might be more realistic target than 10 000 years as a target for our models. However, as most studies are concerned with conservation, it is worth remembering that reserves are generally not set up to maintain a particular number of species, but with more specific conservation aims in mind, and theory may have no relevance to these.

Nevertheless, on a conceptual level, it would seem that elements that MacArthur & Wilson (1967) introduced, such as the role isolation and habitat size can play in predicting species diversity, have maintained their perceived importance, even if not in the way originally envisaged. This view of island biogeographic theory as "a deductive scheme that points out potentially important factors affecting insular communities" (Haila 1990) has become more the norm in recent years. Other theories concerning the avifaunas of habitat patches also tend to have developed from the studies of actual islands. These include the random sampling theory (Connor & McCoy 1979) which suggests that the species composition of an island is a random selection of the fauna. Smaller islands will have fewer species simply as a result of randomly selecting fewer of those available. This theory has been presented as a null hypothesis against which biological explanations may be tested. Coleman *et al.* (1982) supported this theory in their study of breeding birds on islands of a lake, and called it the Theory of Random Placement, stating that the probability of an individual being found on a particular island is proportional to that island's area and is independent of the presence on that island of other individuals of the same species. This is a parsimonious theory but it was not intended to be universally (or even commonly) applicable, rather to be used as a tool to help explore other theories. Nilsson (1986) examined a range of habitats and found that "the results of this study... unequivocally show that communities in small biotope patches are non-random samples from communities in larger patches." Andren (1994) reviewed the effects of habitat fragmentation on birds and mammals and found that the proportion of remaining natural habitat is important in determining whether or not random sampling was valid. He found that where natural habitat made up more than approximately 30% of the landscape, random sampling was a valid hypothesis, because the detrimental effects on fauna were largely due to absolute habitat loss. Below a certain point, however, the effects of fragmentation on species richness and population sizes are greater than those expected from habitat loss alone, as isolation effects and the size of habitat fragments become more important. He also generated

maps simulating habitat fragmentation, which found that above about 60% of natural habitat coverage the landscape wasn't really fragmented (in terms of the natural habitat) because it was distributed in a more continent-like manner.

The third popular theory developed to explain species richness on islands (and by extension on habitat islands) is the habitat diversity hypothesis proposed originally by Williams (1964). This states that larger areas have a greater range of habitats and therefore more niches are available, allowing more species to cohabit. This theory has also found support in a number of studies (Kitchener *et al.* 1982, Rafe *et al.* 1985) and conceptually it is attractive. Buckley (1982) proposed that each island (or pseudo-island) should be broken up into its component habitats and island biogeographic theory applied to these, which would then be summed. This exemplifies some of the problems with this theory, as difficulty may arise in defining what qualifies as a different habitat, and once again the problem of scale crops up. Different taxa will perceive the landscape differently, and so a series of different habitats for one species may be a single habitat for another.

Biological interaction in fragmented habitats is one element that has not received the attention that it perhaps should (extinction in island biogeographic theory is considered to be stochastic). Paton (1994) reviewed a range of studies in North America, and found that both predation and parasitism were mostly higher near the edges of forest patches. Most models of fragmentation would consider these to be part of increased external influences. However, taking the view that the majority of species can and do use more elements of the landscape than a single habitat fragment, they may be considered to be naturally occurring interactions which have increased or decreased in degree. Predation and parasitism from "external" sources are not the only interspecific interactions which may be important in determining the effects of habitat fragmentation. As previously stated, fragmentation may benefit certain already existing elements of the ecosystem while disadvantaging others, altering the balance without the need for any outside influence. Ambuel & Temple (1983) looked at the avifauna of woodlots (3-500 ha) in Wisconsin and found that there were area-dependent changes in interspecific interactions. They suggested that edge and farmland species increase as woodlot area decreases, and that these species then competitively exclude forest-dwelling species, and

that this was more important in determining the avifauna than area-dependent changes in habitat or isolation. This is an approach that merits further consideration.

Ultimately it seems unlikely that any one model can be applied universally to explain and predict the species-area effect. Indeed Wilcox *et al.* (1986) considered that the virtual universality of the species-area effect was because it operated on so many levels, so that at least one of the mechanisms described in the various theories (habitat diversity, stochastic extinction, etc.) would be likely to occur in every case. In support of this Villard *et al.* (1995) found that neotropical migrants in woodlots in Ontario were subject to a combination of passive sampling, vegetation structure preferences, site fidelity, and dispersal depending on isolation from other fragments. Connor & McCoy (1979) stated that habitat diversity or area *per se* (or both) might be the driving forces behind the species-area effect, but the results of either are not qualitatively or quantitatively different. Their review of 100 different studies concluded that there was not one single best-fit model, and that empirical studies were necessary.

1.3 Forest fragmentation in Australia

All natural landscapes undergo some degree of alteration over time, but the rate of alteration has increased as human population and activity has increased. Hobbs & Hopkins (1990) suggested that habitat changes which occurred in the Middle East and Europe as a result of human activities over a period of up to 11 000 years have occurred in Australia over only 200. The long association between human utilisation and landscape in Europe has meant that European ecosystems tend to be resilient, having already lost non-resilient components. Australian ecosystems have had an even longer exposure to Aboriginal land use, especially fire, and can be considered resilient in this regard, but Aborigines remained hunter-gatherers for the duration of their sole occupancy of Australia. European methods of agriculture and animal husbandry cause different types of disturbance, and Australian ecosystems are likely to be more vulnerable to these than European ecosystems.

European settlers sought to impose a European agricultural system on the Australian landscape. This has had a huge effect on both landscape (through clearance of natural

ecosystems) and processes (e.g. suppression of fire and floods). In the Western Australian wheatbelt less than 7% of natural vegetation remains, in a highly fragmented distribution (Saunders & Curry 1990). In addition, the provision of water for stock has also provided water for other animal species. Some bird species have benefited, largely grass feeders which rely on water availability. These may compete with other species. Five bird species have disappeared from the Kellerberrin district in the wheatbelt, and many others are restricted to remnants of natural vegetation (Saunders & Curry 1990). Within agricultural landscapes patches of eucalypt woodland have been left for various reasons, such as provision of shelter or aesthetics. The presence of these patches has given the impression that this habitat type is less at risk of degradation than others, which have received greater attention. The dieback problem currently facing eucalypts in agricultural areas has indicated that this is not the case, and these habitats are becoming more degraded under the current management regime independent of any further clearing (Landsberg *et al.* 1990). Grazing by both livestock and introduced pests such as rabbits can prevent regeneration of native tree species even within native vegetation remnants, and agricultural land is a source of invasive weed seeds (Hobbs 1987). Duncan (1999) stated that in Tasmania "dry sclerophyll forests and woodlands...have suffered greater irreversible disturbance than other forest types."

1.4 The Tasmanian Midlands

The two centuries of European settlement in Tasmania has radically altered the landscape, nowhere more so than in the agricultural region of the Midlands. This lowland area between Launceston and Hobart was originally mostly grassy woodland and grasslands, which were attractive to settlers. The second rural European settlement in Australia occurred in this area. By 1825 most of the grassy lowlands between Hobart and Launceston were occupied by European settlers. Fensham (1989) estimated that over the 175 years following European settlement 83% of the native vegetation of the Midlands had been converted to extensive grazing country with some cropping. European settlement also resulted in a change in fire regime, from Aboriginal "fire-stick farming" to less frequent but more intense fires, which tended to restructure the vegetation from woodland to forest (Duncan 1999). Conversion to agricultural land

continues (Kirkpatrick & Gilfedder 1995) and as a result, the landscape is now dominated by crop, pasture and weed species from Europe.

The Midlands have been subject to different types of land use as defined by Hobbs & Hopkins (1990). The first of these is the complete removal of vegetation and disruption of processes. In the Midlands this has occurred on a small scale with the presence of small towns and some roads. Secondly, there has been replacement of native vegetation with intensively managed ecosystems such as agriculture. This partially disrupts processes and is widespread in those parts of the Midlands suitable for farming. The third land use type is utilisation, which involves exploitation of native vegetation with some modification. This is widespread in the Midlands due to the grazing of stock amongst eucalypt forests and also as a result of timber harvesting, mostly for firewood. Finally, conservation, with minimum deliberate modification of processes is very restricted in the Midlands. Legally protected areas are few in number and extent, but it should be noted that some private land can be and is managed in this way.

Native plant species do survive in this landscape, in rough pasture and some land unsuitable for agriculture, as well as in parks, cemeteries and road/rail verges where native vegetation remains in close to its original state (Kirkpatrick *et al.* 1988). Remnants closely similar to the original vegetation are extremely limited in distribution and extent and continue to be subject to land clearance and degradation. Elsewhere, forested land forms a highly fragmented part of the landscape, existing mostly in isolated remnants of variable habitat integrity. These remnants bear various relationships to truly natural vegetation. Stands of eucalypts amongst agricultural land are subject to influences such as invasion of exotic plant species and grazing which may alter the floristic make-up, structure, or ecological functioning of these stands. Many of these remnants have been incorporated into human agricultural and cultural activity, with grazing and woodcutting both common. Thus, while many remnants may superficially resemble native eucalypt forest, they have been modified directly or indirectly as a result of agricultural activities.

1.5 Thesis Aims

The Tasmanian Midlands presents an ideal opportunity to examine the effects on fauna of habitat fragmentation in an agricultural landscape. The landscape is dotted with an array of remnants of natural and semi-natural vegetation, and these have already been the subject of research into the condition of vegetation and the presence of rare and threatened plants (Kirkpatrick & Gilfedder 1995, Gilfedder & Kirkpatrick 1998). This chapter has introduced the concepts of habitat fragmentation both in global terms and in the context of the agricultural landscape of the Tasmanian Midlands. The next chapter describes the study area and sets out the methods which were used over the whole study (specific methods are included in individual chapters).

The present study aimed to examine the distributions of two groups of fauna, birds and grasshoppers, in a fragmented landscape. Birds have been the subject of considerable research over the years, with particular emphasis on their distribution in fragmented habitats, as discussed above. Tasmanian birds have also received a good deal of attention, and the results have been published in a range of journals, notably the *Tasmanian Bird Report*. However, the Midlands area has received less attention than elsewhere (but see Bosworth 1976). The intention was to investigate the avifauna relatively comprehensively and to compare the results with those found elsewhere in the world and to test the theories outlined above. Thus, the following hypotheses were tested: that measures of species richness and diversity, and the distributions of bird species in eucalypt fragments in the Midlands are the result of (a) random sampling, (b) variability in remnant habitat, (c) the position of the remnant in the landscape, (d) interspecific competition, or (e) a combination of the last three. After a description of general methods in Chapter Two, Chapter Three describes the avifauna of the study area and classifies it into assemblages. These assemblages are tested for differences according to community-level responses such as species richness and diversity, and according to independent variables such as area, isolation and vegetation structure. The trends of the community-level responses are also examined for relationships with the independent variables. The relationships between individual species' presence and/or abundance and independent variables are also examined. The results are discussed in relation to theories regarding avifaunal richness and species' preferences.

Chapter Four attempts to integrate two problems that are prevalent as a result of the conversion of the Midlands into an agricultural landscape; depauperate avifauna and declining tree health. Several factors have been suggested as either proximal or ultimate causes of eucalypt dieback (Heatwole & Lowman 1986), including heavy defoliation by herbivorous insects. The noisy miner, an aggressive honeyeater which excludes most other insectivorous bird species, has been implicated elsewhere in Australia as a cause of both reduced bird species richness and of eucalypt dieback (Loyn 1987, Grey *et al.* 1998). It is hypothesised that tree decline in eucalypt remnants is due to the effects of (a) low rainfall, (b) ageing tree population, (c) presence of noisy miner colonies, or (d) a combination of the above. To this end, the relationships of three measures of tree health with independent variables are examined. The results are discussed in the context of rural tree decline in Australia.

In Chapter Five, community-level responses and occurrences of bird species and of foraging guilds are examined for seasonal trends. The role of habitat fragmentation in affecting these trends is investigated. The hypotheses tested are that migratory and nomadic species are more sensitive to habitat fragmentation, and that small remnants in poor condition have a more temporally stable avifauna than large remnants that are in good condition.

Chapter Six attempts to examine the avifauna on a slightly different scale. Rather than examining entire remnants, community-level responses and species distributions are tested for relationships with independent variables in 200 m segments within the remnants. The larger remnants are also examined for differences in their avifauna between the remnant edges and the interior and differences in vegetation structure are also examined to attempt to explain avifaunal differences. It is hypothesised that the occurrence of open-country bird species is dependent on proximity to open country rather than vegetation structure.

Compared with birds, grasshoppers have been the subject of very few biogeographical studies, and this study intended to provide a basic understanding of habitat preferences of grasshoppers in the Midlands. The variety of invertebrates and difficulty in taxonomy has discouraged biogeographical research, but they have been found to be negatively affected by habitat fragmentation (Didham *et al.* 1996). Further research in the area is

needed, and this study was intended to be a starting point and to provide information on the ecology of an invertebrate group that has been largely neglected. The same hypotheses as outlined above for Chapter Three were tested regarding grasshopper species richness and diversity and grasshopper species' distributions, with the exception that interspecific competition was not considered as a determinant. In Chapter Seven the grasshopper fauna of the study area is described. Grasshopper community-level responses and species distributions are examined for relationships with independent variables as per Chapter Three. Finally, the intention was also that by sampling both grasshoppers and birds from the same sites and by using comparable analytical methods that the results from each group would allow meaningful comparisons to be made between them. The hypothesis was that measures of grasshopper species richness and diversity would not follow the same patterns as that of the avifauna as a result of the biological and ecological differences between the two groups. The discussion considers the impact of habitat fragmentation on this invertebrate group and compares it with the impact on the avifauna.

Chapter Eight considers the results of the present study as a whole and discusses them in both the local and global context. Implications for management of remnants are also discussed.

2 General Methods

A cohesive set of methods was used for the various areas of research presented within this thesis. For this reason it is presented here, and specific methods are presented in individual chapters where appropriate.

2.1 Study Area

This study was a continuation of work on the remnant vegetation in the drier areas of Tasmania (Kirkpatrick & Gilfedder 1995, Gilfedder & Kirkpatrick 1998). All of these remnant and control sites are in the subhumid region of Tasmania, mostly in the Midlands, a lowland graben between the two major cities in Tasmania, Hobart and Launceston. Other sites were in the Derwent Valley, or in the south-east around Hobart (Figure 2.1). The area is approximately 6500 km^2. Apart from some sites in urban situations, the land use surrounding the remnants is extensive farming (mainly sheep) or cropping. The sites are between 41°30' and 43°00' S, and between 10 m and 450 m above sea level.

Remnants of eucalypt woodland or forest had been identified by the previous work but field work proved some of these to be less discrete than was desirable, and a handful were eliminated for this reason. All remnants were at least 25 m distant from other forested areas. With time constraints in mind but wishing to sample as widely as possible (Haila & Hanski (1984) recommended visiting a large number of sites a few times rather than a few sites many times), 46 remnants from the original one hundred were selected. These were chosen to give a range of some of the variables measured by the previous study (particularly area) and also to limit some variables. For this reason site geology was restricted to sandstone, mudstone, sand or dolerite, and dominant eucalypt species were restricted to *Eucalyptus pauciflora, E. viminalis, E. tenuiramis* and *E. amygdalina*, in the hope that larger sample sizes of fewer variables might yield more easily interpreted results. Because *E. pauciflora* sites were less common, an extra remnant dominated by this species but not included in the original one hundred was added to this study.

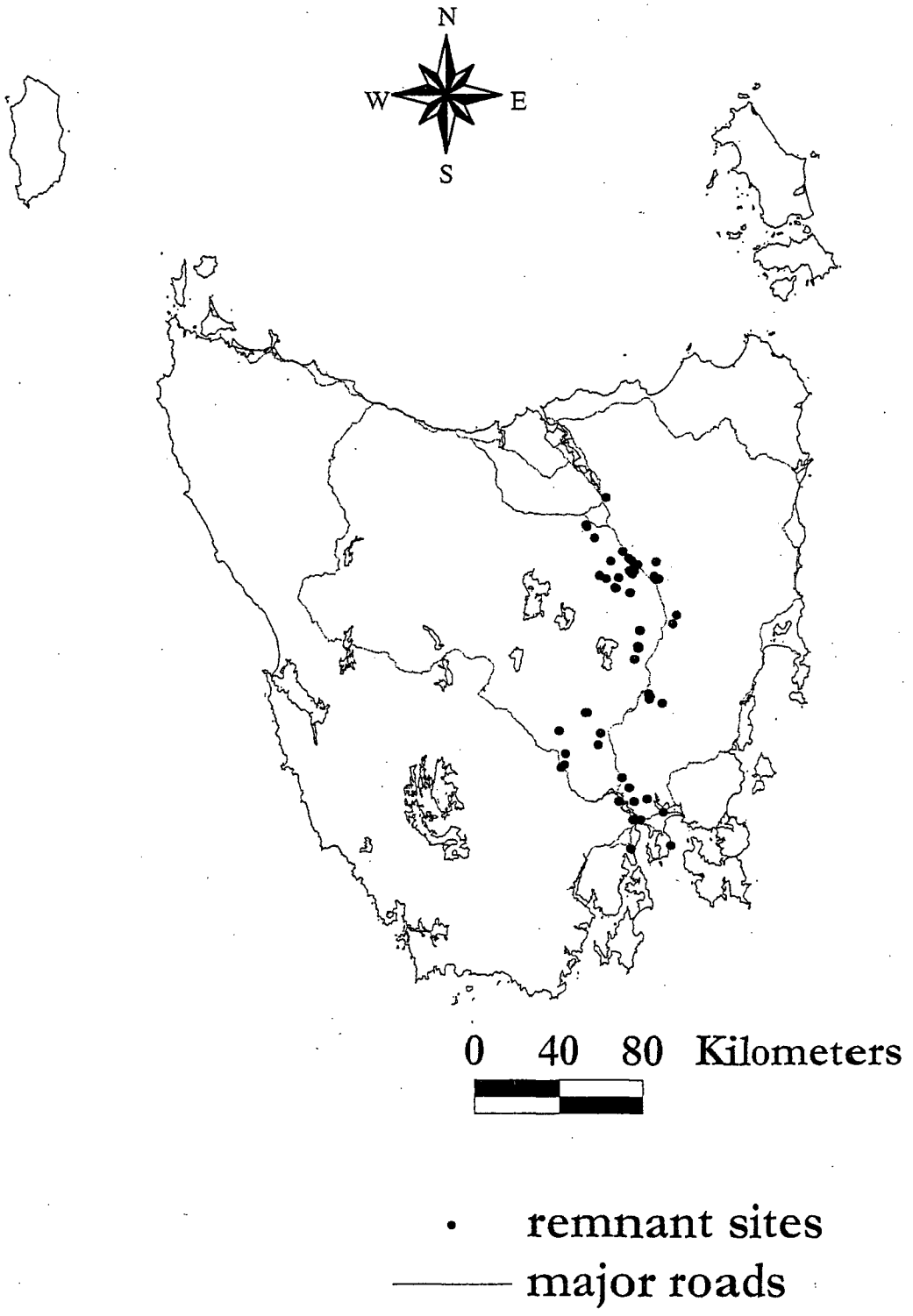


Figure 2.1. Location of remnant study sites

In an attempt to compare the results from smaller remnants with those from larger areas, which might be expected to approximate a more "natural" bird species composition, three larger areas were also selected: an area dominated by *Eucalyptus tenuiramis* near Bothwell; some *E. viminalis* forest near Risdon Brook Reservoir; and the Tom Gibson Nature Reserve. The areas of the latter that were sampled were dominated by *E. amygdalina* or by *E. pauciflora.* The majority of study sites were located on private land, with the exception of some urban parks and the Tom Gibson Nature Reserve. A full list of sites, including grid coordinates is included in Appendix 1.

2.2 Bird and grasshopper surveys

At each site a transect was measured and marked, roughly parallel to the longest axis of the remnant, but also attempting to encounter variation in the habitat, and within the constraints of access to the site. While walking the transects, any bird seen or heard within 50 m was counted, although species flying over transects which were not considered to be using the remnant were not counted. In order to allow comparisons regarding species richness and bird diversity from equal areas, the transects were divided into 200 m segments and it was noted within which segments each bird occurred. Species were identified by sight or by call and named following Watts (1999). The exception to this was that the collared sparrowhawk and brown goshawk were recorded as a single species due to the difficulty in differentiating them consistently in the field. Only species that were not recorded in this study are identified with their scientific names in the text. A full list of scientific and common names of the recorded species is included in Appendix 2.

Conner & Dickson (1980) recommended strip transects as they allow one to cover a lot more area per unit of census time as compared with other methods such as spot mapping. They considered that as long as the counting is consistent then relative values of bird species richness, evenness and abundance should be valid and should allow the use of parametric statistics. Harden *et al.* (1986) reviewed the strip transect method and concluded that vegetation variation between sites may be a problem in comparing counts, but that this problem is common to all methods.

Most studies emphasise the increased activity of birds in the early morning, and half the transects were walked within two hours of sunrise. The other half were walked in the afternoon, to make best use of the time available and to collect as large a data set as possible. In the event the impression was that weather conditions were more important in determining bird activity than was time of day. This bears out the experience of Ratkowsky & Ratkowsky (1979) on Mount Wellington, Tasmania, where they found that wind was a major factor in preventing bird detection. Bird censuses were not conducted in high winds or heavy rain.

The sampling period of this study was two years, from winter 1996 until autumn 1998. Sites were visited in winter, spring and autumn in order to measure seasonal variation. A number of species migrates between Tasmania and mainland Australia, while others migrate altitudinally and/or according to flowering times of significant species. Over the two year period each site was visited twelve times, except where circumstances intervened, these being personal injury or inability to get access to land. The minimum number of visits to any site was ten.

The community-level response variables which were extracted from the bird data for each site were: total species richness, the total number of species recorded from a site over the two year period; mean species richness, the mean number of bird species recorded per visit; local species richness, the average of the mean species number values for each of the 200 m segments into which each transect was divided; percentage dominance, the proportion of all birds at a site which were comprised by the two most abundant birds at that site; total bird density, the mean density of all birds at a site over the two year period expressed as birds per hectare; and species diversity, the Shannon-Wiener diversity index derived from abundance data. The Shannon-Wiener index of diversity (H) combines species number with proportional abundance (i.e. a site with all species of equal abundance is more diverse than one with the same number of species but with most individuals belonging to one species). Magurran (1988) reviewed diversity indices and concluded that measures which only take into account number of species and number of individuals can miss trends in dominance and evenness, so indices which incorporate relative abundance should give a more biologically realistic view of the diversity of sites. The widespread use of this index gives it value in comparing studies.

Grasshoppers were sampled from the same transects as were birds, in the manner described in Chapter Seven. The following community-level responses were derived as described above: total species richness; Shannon-Wiener index of diversity; and percentage dominance by the two most abundant species, while an index of total grasshopper abundance was determined as described in Chapter Seven.

2.3 Independent variables

Data collected by Kirkpatrick & Gilfedder (1995) were used for this study. Additional habitat variables were measured during the course of the present study. The most recent aerial photos of each site were examined to measure landscape variables. These were: remnant area; remnant shape, which was the remnant perimeter (m) divided by remnant area (ha); distance to nearest forest of 50+ ha and of 500+ ha; the amount of forested land within 2.5 km of the centre of the remnant; and an index of local forest fragmentation, which was the mean size of the forest fragments which made up that forested land within 2.5 km of the remnant centre.

Much of the vegetation sampling had already been done (Kirkpatrick & Gilfedder 1995), particularly regarding floristics, and this was not repeated. However, to collect extra information on the structure of the vegetation, particularly in relation to the marked transects, these transects were surveyed for trees. For every tree over two metres tall, within 2.5 m of each transect, species, height, and diameter at breast height (dbh) were recorded. These data were used to derive several habitat variables. These variables were; total stem density (of trees $\geq$ 2m tall) per hectare; stem density in the following dbh classes (<5 cm, 5-15 cm, 15-30 cm, 30-60 cm, >60 cm); live non-eucalypt stems per hectare; live eucalypt stems divided by non-eucalypt stems; maximum eucalypt canopy height; mean eucalypt canopy height; mean canopy height as a percentage of maximum canopy height; total basal area of stems per hectare; basal area of trees divided by number of stems; and an index of vegetation structure, which was the number of stems > 30 cm dbh divided by the number of stems < 15 cm dbh.

For most transects the health of eucalypt species along the transect was recorded according to an ordinal scale ranging from one to six, where one referred to a healthy tree and six referred to a tree with only epicormic shoots. Measures of tree health are discussed further in Chapter Four. Dead trees were also recorded and classified as either eucalypts or other species. One remnant (Queen's Domain) was not surveyed for any vegetation structure or tree health values because a fire near the end of the study period destroyed a large number of the smaller trees along the marked transect.

As noted above, noisy miners aggressively exclude other birds from their colonies and have been implicated as a cause of reduced bird diversity and tree health in studies on mainland Australia (Loyn 1987, Er 1997). The common starling is a significant introduced pest, and it has been suggested that it deprives other tree hollow-nesting species of breeding places (Green 1983). For this reason, abundances of these two species were used as independent variables where appropriate, in order to examine whether these trends were apparent in this study.

Table 2.1 shows a complete list of independent variables used in this study, and notes which of these were collected by Kirkpatrick & Gilfedder (1995). The floristic groups defined by that study were as follows: group one – lowland grassy woodland with *Themeda triandra*-dominated understorey; group two – *Eucalyptus viminalis* or *Allocasuarina verticillata* woodland or open forest with grassy understoreys; group three – *E. amygdalina* or *E. viminalis-E. tenuiramis* grassy woodland and forest on sandstone or mudstone; and group four – dry forest dominated by *E. amygdalina.*

2.4 Data analysis

The aim of statistical analysis was to examine and attempt to quantify the influence of a large number of independent variables (both individually and in concert) on a number of response variables at both the community and species level. Data analysis was performed using Minitab (Minitab Inc. 1997a, 1997b) or DECODA (Minchin 1990).

The Anderson-Darling statistic was used to determine whether variables departed from normality. Where variables were not normally distributed, and where it was statistically

necessary and possible to do so, they were either log-transformed or square root-transformed into normal distributions (as noted in the body of the study). For presentation in the results, they have been back-transformed.

The wide-ranging nature of the present study, combined with the use of variables previously collected, meant that the variables measured were in a range of forms; nominal, ordinal, and continuous (only some of which were normally distributed or able to be transformed into normal distributions). This did not lend itself to the use of stepwise multiple regression, as some of the independent variables would not have been considered in these models. There was also a high degree of correlation between variables. Thus it was decided that for continuous dependent variables (excluding species' abundances) correlation would be employed, and that this would be rank-order correlation where one or both of the variables were not normally distributed. ANOVA or the Kruskal-Wallis test were employed where the dependent variable was continuous and the independent variable was ordinal or nominal. Where significant relationships were established, appropriate statistical tests were used to further examine the nature of these relationships. Understanding of the processes involved in these significant relationships was considered to be important in determining the factors that contributed to the distribution of birds and grasshoppers.

ANOVA and the Kruskal-Wallis test were also used for establishing relationships between bird and grasshopper species distributions and abundances and continuous independent variables. For species' distributions this simply involved presence-absence data, while the abundances of some bird species were divided into three classes to allow the use of these procedures. This method was adopted because it was felt that for species' abundances it provided more information than either logistic regression or rank-order correlation, which were the alternatives (given that abundances of all species were not normally distributed and almost all were not able to be successfully transformed into normal distributions). Whereas the last two methods would have indicated only whether a relationship was significant and in which direction, the use of ANOVA or Kruskal-Wallis tests provides further information regarding the nature of this relationship; whether it is broadly linear, or whether values of the independent variable distinguish between very low abundances of that species and higher ones, or between very high abundances of the species and lower ones. The use of this method

for presence-absence data provides consistency throughout the study and allows valid comparisons to be made between birds and grasshoppers. Chi-squared analysis was used where both variables were nominal or ordinal.

For all analyses, results were considered to be significant if $P \leq 0.05$. Corrections for multiple comparisons were not used, as the probability of a type I error does not differ for any single result whether it is a singleton or part of a matrix of results.

Table 2.1. A complete list of independent variables, including means, standard deviations, minima and maxima for continuous variables.

Name	Mean ± SD	Min.	Max.
<u>Landscape and climatic variables</u>			
Mean annual temperature (°C).*	11.1 ± 0.7	9.3	12.4
Annual mean precipitation (mm).*	617.8 ± 61.0	533.0	823.0
Precipitation in dry quarter (mm).*	124.6 ± 11.6	112.0	171.0
Precipitation in wet quarter (mm).*	180.9 ± 22.5	152.0	244.0
Altitude (m).	216.8 ± 109.4	10.0	450.0
Geology (sand [n=14], laterite [n=9], dolerite [n= 8], sandstone [n=15]).*			
Position in landscape (valley floor [n=12], slope [n=23], hilltop [n=18]).*			
Remnant area (ha)	40.4 ± 38.9	4.0	197.0
Remnant shape (perimeter (m) divided by area (ha))	110.72 ± 42.8	36.0	237.0
Distance (m) to nearest forest area greater than 50 ha.	651.0 ± 997.0	25.0	5275.0
Distance (m) to nearest forest area greater than 500 ha.	1887.0 ± 1744	240.0	8025.0
Area (ha) of forested area within 2.5 km of centre of the remnant.	499.3 ± 304.9	20.8	1640.8
Index of local patchiness (mean size of forest fragments within 2.5 km of remnant centre)	28.0 ± 27.8	4.1	182.3
Nature of surrounding vegetation (pasture [n=48], housing [n=5]).*			
Time since isolation in 1993 (<10 years [n=10], 10-40 years [n=27], >40 years [n=9]).*			
<u>Floristic and vegetation structure variables</u>			
Dominant eucalypt (*E. amygdalina* [n=23], *E. pauciflora* [n=5], *E. tenuiramis* [n=6] *E. viminalis* [n=19]).			
Floristic group (see Kirkpatrick & Gilfedder 1995). 1. n=4, 2. n=7, 3. n=12, 4. n=23.*			
Richness of native plant species.*	63.3 ± 17.2	29.0	109.0
Richness of exotic plant species.*	24.7 ± 9.4	9.0	61.0
Ratio of exotic plant species to native plant species.*	0.40 ± 0.12	0.20	0.67
Significance index for rare plants.*	19.4 ± 20.2	0.0	76.0
Number of stems (of trees higher than 2 m) per hectare.	581.8 ± 554.2	48.2	2768.3
Number of stems per hectare <5 cm diameter at breast height (dbh).	214.9 ± 397.5	0.0	2048.5
Number of stems per hectare 5-15 cm dbh.	198.1 ± 174.0	0.0	629.3
Number of stems per hectare 15-30 cm dbh.	95.4 ± 59.1	0.0	250.8

Number of stems per hectare 30-60 cm dbh.	51.4 ± 27.1	0.0	127.41
Number of stems per hectare >60 cm dbh.	21.4 ± 13.1	0.0	50.73
Index of vegetation structure (stems >30 cm dbh ÷ stems <15 cm dbh)	1.29 ± 2.64	0.02	12.6
Live non-eucalypt stems per hectare.	329.2 ± 505.6	3.9	2545.2
Live eucalypt stems/non-eucalypt stems.	5.1 ± 11.1	0.02	53.0
Live stems 2-6 m tall per hectare.	308.5 ± 489.2	0.0	2335.9
Maximum eucalypt canopy height (m).	21.0 ± 4.7	13.0	33.0
Mean eucalypt canopy height (m).	11.9 ± 3.5	6.1	20.0
Mean eucalypt canopy height as a percentage of maximum canopy height.	56.5 ± 12.8	23.5	90.7
Basal area (cm^2) of trees taller than 2 m (per hectare).	248900 ± 72930	89220	408460
Basal area of trees per hectare divided by number of stems.	851 ± 829	86.0	3832.0
Overstorey recruitment (yes [n=35], no [n=11]).*			
Qualitative assessment of understorey density (open [n=22], medium [n=21], dense [n=10])			
Percentage of trees removed by logging.*	11.8 ± 12.5	0.0	48.0
Tree health variables			
Mean dieback score (where more than ten eucalypts were measured).	3.32 ± 0.79	1.8	5.3
Percentage of eucalypts with epicormic shoots.	31.1 ± 17.5	2.2	67.6
Percentage of dead stems of all species.	18.0 ± 11.3	0.0	39.3
Management variables			
Stock rate (none [n=17], light [n=14], medium/heavy [n=28]).*			
Use of fire as a management tool (yes [n=10], no [n=35]).*			
Bird species variables[a]			
Noisy miner abundance (birds/ha)	1.15 ± 1.03	0.00	3.72
Common starling abundance (birds/ha)	1.21 ± 1.23	0.03	5.67

* variable measured by Kirkpatrick & Gilfedder (1995). Refer to their work for methods of determination.

[a] these variables were not used for analysis in cases where they directly affected the results (e.g. differences in bird assemblages)

3. Birds of eucalypt remnants in Tasmania's Midlands

3.1 Introduction

3.1.1 Birds in fragmented forests

Studies of forest fragmentation and its effects on birds, are many and varied, although there is a preponderance of work in the temperate regions of Europe, North America and Australia. The latter continents have proved especially fruitful, possibly because of the relative speed with which the fragmentation has occurred. Almost all research has found that forest fragmentation is detrimental to avifaunal conservation. However, the aims and results of studies differ greatly. The role of landscape features such as area and isolation in explaining avifauna diversity and bird abundance has been a common theme, although it has become increasingly apparent that species richness or diversity *per se* may not be the best indicators of either habitat quality or bird community health. The value of species-level responses has also become increasingly appreciated over time. An increase in species number with increasing habitat area is a widespread phenomenon, but not necessarily important, as larger sample sizes within the one habitat will also give an increase in species numbers (van Dorp & Opdam 1987). Measures of diversity have indicated a larger number of species in large forest patches, but the number of species per unit of habitat area may also be important. Lynch & Whigham (1984) found that small forests had more species at a given point. However, this does not mean that a small forest is more diverse. The significance of finding more species per unit area in a small forest, which overall has fewer species than a large one, is that the small forest is likely to be more homogeneous in terms of its avifauna. The species composition of any area is also of vital importance.

Studies of forest fragmentation in Europe might be expected to find relatively fewer connections with bird species distributions and bird community values, on account of the longer time frame of habitat modification. However, this does not appear to be the case. A series of studies of small deciduous woodlots (0.1-39 ha) in an agricultural

landscape in Holland (Opdam *et al.* 1984, 1985, van Dorp & Opdam 1987) found that patch area and isolation, and regional forest cover, were the factors which best explained the distributions of those bird species which they deemed to be forest-dwellers. Different species responded differently to changes in area and isolation, although 26 of 32 species showed a significant relationship with forest patch size, even if only a few species could be realistically considered to be inhabiting true habitat islands. Species in this category were generally restricted either by habitat territory requirements or by behaviour which prevented them using external habitat, or by a combination of both. Van Dorp & Opdam (1987) disputed the findings of Helliwell (1976), who considered that isolation was not an important factor in British woods, on the basis that his study sites were too heterogeneous, and they found that the distribution of several species in their study was related to isolation and connectivity indices. The number of forest-interior species was significantly affected by the density of connecting elements.

Elsewhere in Britain studies have found results more in keeping with those found in Holland. McCollin (1993) found that between-patch factors such as area and isolation were more important than within-patch variation such as vegetation structure in explaining patterns of abundance and species richness in woods. He divided bird species into three groups: a group of woodland-dependent species which decreased in frequency with increasing fragmentation; a group of edge species which increased with increasing fragmentation, and whose abundance was possibly more closely linked to resources in surrounding habitats than any characteristics of the woodland site; and a group of ubiquitous species which were widespread but affected by fragmentation on a local scale, and which possibly had their populations buffered by numbers from surrounding sub-optimal habitat. The identification of species as either insensitive (seemingly) or responsive to patch characteristics, particularly area, was also made by Ford (1987). Bellamy *et al.* (1996) found that patch area explained around 70% of the variation in breeding woodland species numbers, although edge species numbers were generally not related to patch size, with perimeter length a more significant variable for the species.

In other parts of Europe the story is similar. Bird species numbers on remnant oak-hazel sites (0.2-12 ha) in southern Sweden were mostly related to area. Adjoining

habitats were not important, even when these remnants were adjoined by conifer forest as opposed to open country (Hansson 1997). In Spain, Telleria and Santos (1995) hypothesised that as remnant size diminished, birds with similar habitat preferences tended to disappear simultaneously, meaning that the remnants displayed a nested pattern of species distribution. Their data supported this hypothesis, and they suggested that this reflected the fact that birds utilising relatively scarce resources were most likely to disappear when habitat area was reduced, and that habitat specialists requiring large areas of contiguous habitat were the main victims of fragmentation. Associated with this idea is the concept of habitat thresholds, at which species diversity and/or species composition dramatically alters. These potentially occur for any habitat variable, such as resource density or competition levels, but in practice they have been applied to habitat size, as it is more difficult to establish thresholds for other variables. Cieslak & Dombrovski (1993) looked at small (0.04-15 ha) woods in Poland, and suggested that there was a series of thresholds, at which the avifauna altered in different ways. For example, area was not important until forest size reached 0.4 ha, and from previous data they proposed another threshold at around 25 ha, at which bird species richness begins to climb more steeply. The threshold concept has been supported in North America, where Blake & Karr (1984) in Illinois also found evidence of patch area thresholds, as forest-interior migrants were almost never found in forests less than 24 ha in size.

Large-scale forest clearance has occurred more rapidly in America and Australia than in Europe, with radical landscape changes happening over decades rather than centuries. Because of the differences in time scale and also because of the apparent importance of species-level responses, one might expect to find some different trends throughout the world. In North America the results are influenced by the large number of neotropical migrants, which return to temperate forests to breed and which, as a group, tend to be more sensitive to fragmentation than resident species. The sensitivity of this group is generally independent of the influence of isolation, which is not surprising, considering the distances travelled by many species in the course of migration.

Blake & Karr (1984, 1987) looked at forest patches ranging from 1.8 to 600 ha in Illinois and found that bird species richness was influenced by forest area rather than isolation, although when the avifauna was divided into ecological groups habitat variables were most important for short-distance migrants. However, when individual species'

abundances were considered, habitat variables correlated with over two thirds, compared with less than half which correlated with area. This exemplifies the importance of looking beyond mere species numbers and examining the responses of individual species. In terms of the species compositions of small forests, they found that they were not random assemblages but a subset of those in larger forests. In conservation terms these results are important, because although this study found that two smaller reserves were likely to support a greater species number than a single large one, long-distance migrants and forest-interior birds were poorly represented in the smaller forests. These species are more likely to require protection, and the importance of considering specific aims in conservation cannot be over-emphasised.

The trend of long-distance forest-dwelling migrants increasing the bird species richness of large forests is supported elsewhere in North America, including Wisconsin (Ambuel & Temple 1983), the mid-Atlantic states (Robbins 1980, Robbins *et al.* 1989), Maryland (Whitcomb *et al.* 1981), and the north-eastern USA (Askins *et al.* 1987). The last study suggested reasons for the decline of forest-interior birds in the face of fragmentation: greater susceptibility to edge effects, including predation and parasitism; absence of microhabitats from small forests; minimum territory requirements may not be met; higher rate of stochastic extinction due to small population size; and small forests may be different from large in vegetation and microclimate. In eastern North American deciduous forest Whitcomb *et al.* (1981) found that only 19 of 93 species could be considered as inhabiting true habitat islands. Of the others, some had large territories combining more than one fragment, while others were not really forest species, rather they used the intervening habitat more than the forest. Species intolerant of fragmentation tended to be highly migratory, specialised for forest interior habitat, build open nests and to nest on the ground.

Lynch & Whigham's (1984) study of 270 patches of forest in Maryland examined the importance of various environmental factors for bird species as well as measures of bird diversity. The local abundance of all common bird species was affected by one or more of the following factors: size, isolation, floristics, physiognomy and successional maturity, but area was significant for only 8 of 31 species, and most important for only three of these, whereas isolation, plant species diversity and tree physiognomy were important for a much wider range of species. The only correlate for bird species

diversity (Shannon-Wiener diversity index) was a negative one with tree density, while bird species richness was negatively correlated with size (signifying that small patches had more species at a given point). They argued that area has been over-emphasised as a determinant of species diversity in other studies because other factors have not been thoroughly examined, and because area correlates with many other environmental factors, meaning that the strong correlations between area and species diversity may be an artefact. In Ontario, Freemark & Merriam (1986) also disputed the pre-eminence of area effects, finding that habitat heterogeneity (derived from forest structure and spatial variability) correlated best with bird species richness, although area was important for long-distance migrants. Bird density and diversity (measure not stated) of wintering bird populations in Tennessee were also found not to be area-related (Hamel *et al.* 1993). Lynch & Whigham's (1984) findings were questioned by Robbins *et al.* (1989) who felt that they did not look at patches at the smallest end of the scale and that their definition of isolation was not appropriate. Despite their findings, Lynch & Whigham (1984) felt compelled to discuss the importance of habitat size, making the interesting point that "above some critical minimum patch area, floristically and physiognomically 'rich' forests may support bird communities that are quantitatively and qualitatively similar to those found in larger, but 'poorer' forests."

Ambuel & Temple (1983) introduced the concept of fragmentation influencing species interactions. They ascribe the changes in bird diversity in small woodlots to increased competition from edge and farmland species as woodlot size decreased rather than any habitat changes resulting from fragmentation. This idea of interspecific interaction has been examined in other studies, but largely in terms of predation (particularly of nests) and parasitism rather than competition. Paton (1994) reviewed a range of studies in North America, and found that both predation and parasitism were mostly higher near the edges of forest patches, and that there was a positive relationship between nest success and forest patch size. He suggested that the limit of these edge effects was probably 50 m rather than 100-200 m as suggested in other studies. Donovan *et al.* (1997) supported this idea generally, but argued that the effect of edge habitats is not the same in all landscapes, with nest predation and parasitism levels differing depending on the level of fragmentation in the landscape. This corresponds with work by Andren & Angelstam (1988) in Sweden, who found that nest predation was higher in farmland and edge habitats than in forest interiors in a landscape comprising 58% farmland, while

an earlier study in a landscape comprising 3% farmland did not find these trends. This implies that the landscape level of fragmentation can affect the impact of the presence of habitat edges and coincides with the theoretical work of Andren (1994) as discussed above. The nature of the surrounding habitat(s) can also be important, and Tewksbury *et al.* (1998) found that nest predation was lower in fragmented landscapes than in forested landscapes in an area of Montana that was prone to natural disturbance. This was probably because of lower abundance of forest predators not being compensated for by higher abundance of open-country predators. Janzen (1983) suggested that patches of natural habitat may remain ecologically intact longer if surrounded by grazed land than if surrounded by secondary successional land, which may be rich in plants and animals which can invade the pristine forest. One would also expect that edge effects would vary in extent of penetration according to the effect in question.

Further evidence of altered bird species interactions as a result of forest fragmentation comes from studies of the brown-headed cowbird (*Molethrus ater*) in north-eastern North America (Mayfield 1977). This species, which is the only obligate parasite in North America, prefers small forest areas among grassland, and prior to European settlement it was found in these habitats west of the Mississippi. Widespread transformation of the north-eastern forests into farmland provided ideal habitat for the cowbird, and it has since spread throughout this region. In these new areas it has been implicated as a significant cause of reduced breeding success among songbirds, which had not adapted to the threat of parasitism presented by the cowbird and which were more susceptible to it. Robinson *et al.* (1995) found that nest predation and parasitism by cowbirds increased with forest fragmentation in the mid-west of the United States, in several landscapes which varied in forest cover from 6% to 95% (within 10 km of study sites). The increased abundance of the brown-headed cowbird was particularly damaging to the endangered Kirtland's warbler (*Dendroica kirtlandii*), with parasitism rates of up to 70% (Mayfield 1977). Control of the cowbird starting in 1972 led to an increase in warbler reproduction. This change of competition and interaction has often been overlooked in the reduction of bird numbers and species.

3.1.2 The Australian situation

Australian studies have mostly been in agreement with findings overseas. Indeed Howe (1984) compared small forest patches (0.1-7 ha) in NSW and Wisconsin as well as controls in large contiguous forest areas, and found that birds showed common patterns of distribution, even though the two locations differed considerably in taxonomy, seasonality and habitat. Area was the best predictor of species richness, and assemblages within the small patches were more predictable over space and time than those in the controls. In the New England area of Australia, while looking at a wider range of patch sizes (6-400+ ha), Barrett *et al.* (1994) found that only 17 of 131 species were restricted to patches over 400 ha in size. For most species habitat quality was at least as important as patch size, even though bird species richness was greater in large remnants. They felt this was likely to be due to better habitat quality in larger remnants. Also in northern NSW, McIntyre and Barrett (1992) concluded that only five (of around 55) species were truly living in a fragmented habitat, and were restricted to fragments of 200 ha or more, while the other species used the entire landscape to differing extents.

The wheatbelt of Western Australia has been the subject of a number of studies of habitat fragmentation, which is understandable given that only 7% of the area remains naturally vegetated as a result of land clearance for agricultural purposes. Kitchener *et al.* (1982) looked at remnants ranging from 38 to 5119 ha, and felt that the effects of area overrode other habitat effects, particularly for small remnants, while isolation was not a significant factor. The presence of bird species in remnants of less than 600 ha did not correlate with plant structure or floristics, while the presence of those in larger remnants did, evidence that smaller remnants inadequately sample habitat diversity. They considered that reserves as small as 80 ha could be important for conservation purposes, but that a series of reserves of 1500 ha of natural habitat was required to maintain avifauna in the long term. No single reserve contained more than 71% of the region's bird species, and their extrapolations suggested implausibly large areas to contain the whole avifauna in one reserve. This would seem to provide evidence for the benefits of several smaller reserves if the overlap of species between reserves is not great. Arnold & Weeldenburg (1998) looked at smaller remnants (less than 27 ha) of one eucalypt species, *Eucalyptus salubris*, and found that small passerines were less frequently found in small remnants. Most large birds were more commonly found in small remnants, and

structure was not significant for these species. All the small passerines were associated to some extent with forest structure indices, which implies that the smaller remnants are likely to be more degraded. This introduces the idea of habitat quality and its relationship with forest size. Barrett *et al.* (1994), in a study of the New England region of New South Wales, considered that as the larger patches tended to be in better condition, it is possible that some of the birds restricted to these are sensitive to disturbance rather than size. Van Dorp & Opdam (1987) found weak correlation between patch size and habitat characteristics in Holland, and suggested that it is at least possible that differences in habitat quality may be masked by patch size to some degree.

By contrast, Catterall *et al.* (1997) found little fragmentation effect on wintering birds down to around 10 ha in eucalypt forest in south-east Queensland, and they suggested that habitat loss rather than fragmentation is likely to be the main cause of regional declines in forest winter migrant birds. The smallest remnants had fewer forest-dependent species, and increased abundance of noisy miners and Australian magpies. This study supported the concept of habitat size thresholds, in this case of around 10 ha, below which a significant proportion of forest species are not present, and they suggested that interference competition from noisy miners may contribute to this. This threshold concurred with work by Loyn (1987) in Victorian forests. Combinations of small forest patches had about the same number of species, possibly more, than large ones of the same area, because of the presence of farmland species. But the numbers of individuals of forest species in small patch combinations was smaller, and mostly consisted of transients except where understorey remained. Fragments smaller than 10 ha had mostly been heavily grazed and had little understorey, and starlings, magpies, eastern rosellas, magpie-larks and noisy miners were dominant. This indicated that the major effect of forest clearing has been to reduce and deteriorate habitat rather than to increase theoretical fragmentation effects such as isolation. Robbins *et al.* (1989) noted that in the mid-Atlantic states of the USA, "...in relatively undisturbed mature forests studied, degree of isolation and area were significant predictors of relative abundance for more bird species than were any habitat variables." The distinction between habitat area and fragmentation effects, and disturbance/degradation effects associated with habitat loss seems problematic.

In summary, the trend throughout the world has been that increased area of habitat results in not only more species, but greater bird diversity (generally measured as the Shannon-Wiener Index). However, species per unit area and bird density have not shown the same trends, in some cases tending the opposite way. Studies which have examined the responses of individual species have found that only certain species have distributions which correlate significantly with area and have emphasised the importance of looking at species-level responses to habitat variables. In some cases the primacy of area in explaining bird community characteristics has led to ignoring other factors with potential to explain other trends.

3.1.3 Avifauna of the Tasmanian Midlands

Tasmania has an impoverished avifauna when compared with mainland Australia. Ridpath & Moreau (1966) consider that even fewer species were present on the island at the height of the last glacial period, 18 000 ago. They felt that the majority of species common to sclerophyll forest and savanna woodland habitats would have been totally or virtually absent from the island due to a lack of suitable habitat. Since that time immigration has occurred, particularly while Tasmania was connected to the mainland. Thomas (1974) noted that Tasmania had 104 resident land bird species, as compared with 285 for Victoria, the Victorian total becoming 176 if those living in habitats absent from Tasmania were excluded. He concluded that Tasmanian habitats are saturated, and therefore at equilibrium, because the number of species at a given point is similar between habitats, both in Tasmania, and on the mainland. Nine exotic species have become established in Tasmania since European settlement: the laughing kookaburra, superb lyrebird (*Menura novaehollandiae*), goldfinch, greenfinch (*Carduelis chloris*), house sparrow, common starling, blackbird, skylark (*Alauda arvensis*), and spotted turtle-dove (*Streptopelia chinensis*). At least one further species, the cattle egret (*Ardea ibis*), has recently naturally expanded its range into Tasmania. All of these species, except the superb lyrebird, are associated with European cultural landscapes. Only one species, the Tasmanian emu (*Dromaius novaehollandiae diemensis*), is known to have become extinct on the island since European settlement.

Precise details of the avifauna of the Midlands during the early days of European settlement are not easy to come by. Legge (1902) gave a brief description of Tasmania's birds, and mentioned the presence of parakeets, honeyeaters, pardalotes and thornbills in the open country of the Midlands. It is reasonable to assume that all the native species now present were also present 200 years ago, and there is no evidence that any species apart from the Tasmanian emu has disappeared from the area. However, the relative abundances of species, and their ecological roles are far harder to establish. Given the suitability of certain species, such as Australian magpie and forest raven, to the agricultural and urban landscapes established following European settlement, it is reasonable to question how widespread and abundant they were previously. Answers to such questions are hard to come by, but we can gain some insight by examination of sites that approximate more closely the original state of the vegetation of the Midlands. For this reason, observations in "control" sites of large areas of good quality habitat have been included in the present study.

In more recent times, Bosworth (1976) recorded 50 species in his study of birds in habitat islands of dry sclerophyll forest in Tasmania. This study of only eight sites (two of which were excluded from analysis) and only three independent variables (area, isolation and elevation) took a strict island biogeographic view and came to the not surprising conclusion that area best explained the number of species present. They did note the aggressive behaviour of noisy miners at one site, and suggested their presence may have helped to explain the absence of small birds from that site. Ratkowsky (1983) recorded 55 species on Mount Wellington on both dry and wet sclerophyll sites, ranging from 240 to 1270 masl, between September and April. Taylor *et al.* (1997) recorded 55 species over seven years in dry sclerophyll forest in south-eastern Tasmania, some of which was regrowth.

Species records (Thomas 1979) and personal observation show that the Midlands also supports a number of non-forest species, particularly wetland species, including several ducks, grebes and egrets, which are variably widespread and abundant. In addition, a number of species more commonly associated with wet forests may be encountered, either due to the presence of suitable microclimate or because of unusual movement of these species. These include the yellow-tailed black cockatoo, white-browed scrubwren and olive whistler. The *Tasmanian Bird Report* issues from the 1970s describe most

species as occurring in roughly the same distributions as today. Newman (1983) noted that the eastern rosella, amongst other species, had been described as on the decline due to competition from starlings for nesting sites, but that this was not the case. He noted that "it appears eastern rosellas often breed within noisy miner colonies, where starlings are either scarce or absent." Recent anecdotal evidence has suggested that noisy miners are on the increase, and many landowners have noticed their aggressive behaviour.

The major division in distribution of forest bird species in Tasmania is between those of wet forests and those of dry forests. No species is restricted to alpine areas, and altitudinal migration is characteristic of several species. No species is entirely or mostly limited to the dry forests and woodlands of the Midlands. The distribution of those species which prefer forest and woodland habitat must necessarily have been reduced, although the agricultural landscape has benefited some species, such as the sulphur-crested cockatoo and the grey butcherbird. The presence of exotic species is not as marked as the presence of exotic plant species, although several species are present and possibly increasing in distribution and numbers, including the laughing kookaburra, goldfinch, house sparrow and common starling, which is particularly widespread and abundant. Anecdotal evidence suggests that the relative abundance of species has altered dramatically in this highly fragmented landscape. Species which prefer (or are better able to adapt to) disturbed or edge habitats seem to be on the increase. These include the noisy miner, Australian magpie, and grey butcherbird. Species that require more extensively forested habitat are on the decline. Disturbed and edge habitats were undoubtedly a feature of pre-European landscapes. However, the proportion of forested areas that they make up seems to have significantly increased in the last two centuries.

Bird (and other animal) communities have not been described in Tasmania, partly because this process seems less appropriate than for plants, and partly because of a lack of systematic description of bird associations. However the fact that the avifauna seems to be changing, while no individual species is threatened with extinction may make this a valid approach. The major consideration for conservation purposes in the study area appears to be the maintenance of a diverse suite of bird species rather than concern for any one species. Two threatened species, the forty-spotted pardalote and the swift parrot, do inhabit dry sclerophyll forest, but the former was not recorded within the

study area, and the latter only incidentally, as it prefers *Eucalyptus globulus* forest. The former has had a restricted distribution for its recorded history, and seems to have suffered from competition by closely related species (Woinarski 1985). However, the extensive nature of land clearance in the Midlands must have reduced the populations of species that lived in the woodlands and forests which formerly covered this area, even if none of them seem at immediate risk of extinction. Agricultural areas of Australia have a beleaguered appearance in the present day, with economic and other environmental issues stacked against them. The diminishment of their avifaunas may seem the least of their worries; however, this does not reduce the ecological value of healthy bird communities, and their economic value may have been underestimated.

The intention was to examine the impact of habitat fragmentation on birds in an agricultural and urban landscape in Tasmania. To this end, the relationships between a range of bird community-level responses and independent variables (climatic, landscape, vegetation, floristic, management and competitive) were investigated. In addition, the relationships between these independent variables and the distribution and abundance of individual bird species recorded were examined, in order to determine whether species displayed different trends from measures of bird species diversity, richness and richness. It was hypothesised that bird species richness and the distributions of bird species in eucalypt remnants are the result of (a) random sampling, (b) variability in remnant habitat, (c) the position of the remnant in the landscape, (d) interspecific competition, or (e) a combination of the last three.

3.2 Methods

3.2.1 Bird Assemblages

Ward's method was used to agglomeratively classify the avifauna into assemblages on the basis of presence-absence data. It was decided to use presence-absence rather than abundance data as the latter tended to be less reliable, and because single flocks of birds spending a short time on a site could distort the classification, whereas presence-absence data were not distorted by this. Abundance data were used for other analyses.

Non-metric multidimensional scaling, using the default options in DECODA (Minchin 1990) was used to place sites along orthogonal axes which best explain their avifaunal differences. This was done using abundance data.

To avoid confusion, the term "assemblage" is used when referring to the classificatory groups derived. Community-level responses refer to qualities of species richness and diversity at each site.

Analyses of variance were used to detect differences in characteristics of the derived bird assemblages. ANOVA was also used to test for differences between distributions of bird assemblages on continuous variables. Chi-squared analysis was used to test the significance of differences between expected and observed values for bird assemblages against ordinal variables.

3.2.2 Community-level response variables

Product moment correlation coefficients were used to examine relationships between continuous independent variables and the bird community characteristics. ANOVA was used to examine the relationships between variables in ordinal classes and bird community characteristics. Where necessary, variables were log- or square root-transformed to obtain a normal distribution. Tukey's method was used to detect which means differed within the ANOVA. Where transformation was not possible, rank order correlations or Kruskal-Wallis tests were performed to examine relationships between independent and response variables. In practice this was rarely necessary, and almost all cases in which they were used non-significant results were obtained.

3.2.3 Bird species' distributions

The habitat preferences of the 37 species that occurred at more than 10 sites were examined. For species' presence and absence at sites, chi-squared analysis was used to test the significance of differences between expected and observed values of variables

recorded at an ordinal scale. ANOVA and Kruskal-Wallis tests were used to test the significance of differences in values of continuous variables. Four species were too widely distributed for these tests to be valid for presence-absence data. These species were placed into three abundance classes; low, medium, and high (exact parameters chosen by examining data and varying from species to species) and the same analysis techniques as for other species were applied. This was also done (in addition to presence-absence) for 18 other species, to examine whether abundance showed different (or more complex) trends than simple presence/absence.

Experience in the field and preliminary data analysis showed that the presence of noisy miner colonies had a significant influence on measures of bird species richness, diversity and density. Many bird species were excluded from these colonies. To examine trends in community level variables and species distributions, the same analyses as described above were performed on the subset of remnants not dominated by noisy miner colonies.

The reduced sample size as a result of excluding remnants dominated by noisy miner colonies violated statistical assumptions in some cases, and the following alterations were made as a result. Overstorey recruitment was not used for these calculations, as only three sites had no overstorey recruitment. Sand and sandstone were amalgamated into one geological class, *Eucalyptus pauciflora* and *E. tenuiramis* were amalgamated as dominant eucalypt class, and floristic group one was not included for these analyses, in order to remove small sample sizes for these categories. In order to investigate whether these alterations affected the results already gained for the complete data set, analysis was performed using all sites with the alterations described. No extra significant results were obtained and any significant results previously established were still valid.

3.3 Results

Sixty-two taxa of bird were recorded within the remnants and controls. These included seven endemic and five exotic species. Table 3.1 shows means, maxima and minima for measures of species richness and bird density. Forty-two species were recorded from at

least 10% of sites. The common starling was the only species to be recorded from every site, and it was also the most abundant species over the whole study. Other species which were particularly widespread were the forest raven, noisy miner, striated pardalote, grey butcherbird, Australian magpie, laughing kookaburra, green rosella and eastern rosella. This distribution was not always reflected in abundance; species such as the grey butcherbird and forest raven were present in most remnants but only in small numbers. By contrast, the brown thornbill was more restricted in its distribution but more abundant where it was present, and it was the third most commonly recorded species, after the common starling and noisy miner. Apart from the brown thornbill and striated pardalote, the birds mentioned above are considered edge species or birds of open country and farmland. Species more commonly associated with forests, such as the grey fantail and golden whistler, tended to be more limited in distribution, although they could be abundant when present.

Table 3.1. Summary statistics for remnant species richness and bird density

	Minimum	Maximum	Mean
Total species richness	7	36	21.49
Mean species richness	3.75	18.33	9.35
Local species richness	2.9	9.2	5.1
Total bird density (birds per hectare)	3.65	17.76	7.91

3.3.1 Classification

Examination of the classification dendrogram (Figure 3.1) and of raw data led to a decision to divide the avifauna of the sites into four assemblages. The first division (between assemblage one and the other three assemblages) was considerably more pronounced than any of the others, which reflected the situation in the field. Ordination confirmed these groupings (Figure 3.2). For further analysis, in some cases the higher level of division (into two assemblages) was used so as not to violate assumptions of statistical methods. This is noted where it occurs.

Assemblage One was present in 23 remnants, as well as one of the transects located in control areas. In effect, this assemblage was found on remnants that were entirely dominated by noisy miner colonies. This assemblage was dominated by "edge" species, and was distinguished by high presence values for the grey butcherbird, Australian magpie, eastern rosella, laughing kookaburra and noisy miner. Several species more commonly associated with forest habitats were largely or entirely absent. These included the brown thornbill, yellow wattlebird, dusky woodswallow, grey shrike-thrush, yellow-throated honeyeater, golden whistler, spotted pardalote, crescent honeyeater and superb fairy-wren.

Assemblage Two: was present in 12 remnants and one control transect. While most of the species in the assemblage one were also present in assemblage two, they were less dominant, being mostly confined to the edges of larger remnants or occurring infrequently. Instead, a suite of small insectivorous birds were commonly present – thornbills, honeyeaters, pardalotes, grey fantail and others. Species distinguishing Assemblage Two included the dusky woodswallow, black-faced cuckoo-shrike and striated pardalote.

Assemblage Three: was found in five remnants and one control transect. It bore some similarity to assemblage two, but "edge" species were less frequent or totally absent. These included the noisy miner, eastern rosella, Australian magpie and grey butcherbird. The above-mentioned group of smaller insectivorous birds was most commonly encountered. Species which distinguished this assemblage included the fan-tailed cuckoo, dusky robin and flame robin.

Assemblage Four: was found in seven remnants and three control transects. This assemblage presents a less cohesive picture than the others do. Several of the remnants in which it occurred were urban or semi-urban parks. Species most commonly recorded in this assemblage included the swift parrot, eastern spinebill and house sparrow. Most "edge" species were uncommonly present in these remnants, as were some "forest" species, such as the green rosella and common bronzewing.

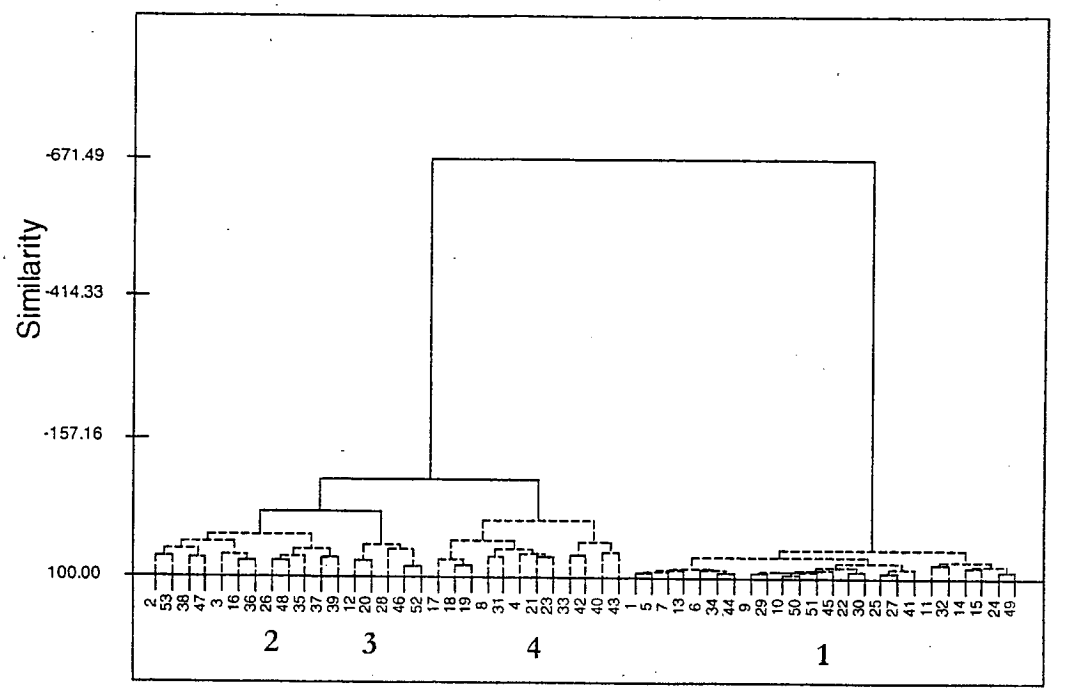


Figure 3.1. Classification of remnants into avifaunal assemblages
Dashed lines connect sites with the same bird assemblage (numbers refer to bird assemblages described in the text)

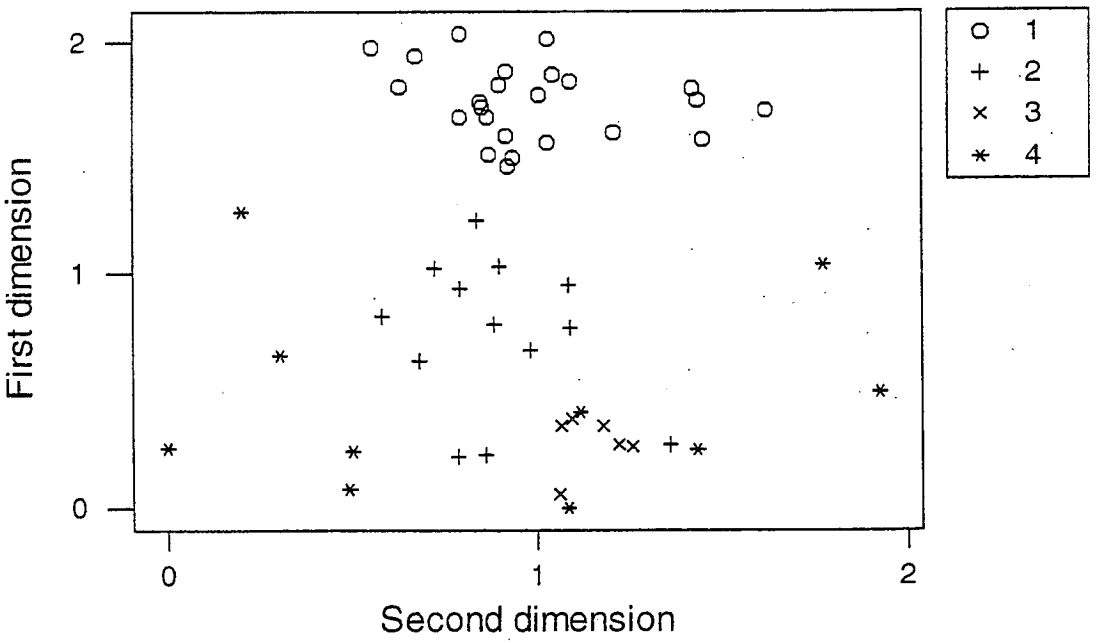


Figure 3.2. Two-dimensional ordination of remnants according to frequency of bird species' distributions
Symbols refer to the bird assemblages defined by Figure 3.1 and described in the text

To summarise, Assemblage One (remnants dominated by noisy miner colonies) was markedly different from the others. Classification, ordination and observations in the field suggest that there is not a continuum between Assemblage One and the others, rather a considerable gulf. By contrast, the other assemblages are less discrete entities.

3.3.2 Bird assemblages, independent variables and community-level responses

Table 3.2 shows the expected and observed results for those independent variables which differed significantly in Chi-squared analysis. Table 3.3 displays a list of values of continuous independent variables and of community-level responses for each of the assemblages and notes any significant differences.

Bird assemblages differed markedly in their values for community-level responses. Assemblage one had significantly lower values for mean species richness, total species richness and species per hectare than did all other assemblages. Assemblage three had higher values for these variables than did all the other assemblages. Dominance of the two most abundant species was greater for assemblage one than for all other assemblages. Total bird density was significantly higher for assemblage three than for assemblage one.

Remnant area and shape were the only landscape variables to distinguish the assemblages. Measures of isolation and local forest cover showed no relationship with bird assemblage distribution. The relationship between remnant area and assemblage distribution is a very strong one. Of 16 remnants smaller than 20 ha, only three did not support assemblage one, and two of these had very dense understoreys (*Beyeria viscosa* in one case, *Ulex europaeus* in the other). Of 23 remnants larger than 30 ha, all but five supported assemblages other than assemblage one. Those that supported assemblage one had very open understoreys. Of the eight remnants 20-30 ha in size five supported assemblage one and three did not, and again understorey density was useful in explaining the differences.

Assemblages One and Four differed on several measures of vegetation structure. These were: overall stem density; density of stems <5 cm dbh and 5-15 cm dbh; index of

vegetation structure; basal area divided by number of stems; density of trees 6 m or shorter; and density of non-eucalypts. Assemblages Two and Four also differed on these last two variables. Measures of canopy height showed no relationship with bird assemblage distribution.

Measures of eucalypt health (scores for dieback and dead eucalypts) were significantly poorer for Assemblage One than for Assemblages Three and Four. Assemblage Two had a significantly higher percentage of dead stems. Assemblage One had higher levels of logging than did Assemblages Two and Four.

Assemblage Four had significantly higher values for both exotic vascular plant species richness and native vascular plant species richness than did all other assemblages.

Assemblage Three was located on sites with significantly higher mean annual precipitation than Assemblages One and Two, and this trend also showed for precipitation in the wettest quarter. Assemblage Four was also found on sites with higher mean annual precipitation than Assemblage Two.

Chi-squared analysis showed that assemblage one was not found where the surrounding landscape was urban. Assemblage One was also found less frequently than expected where the understorey was dense, and more frequently where it was open. The other assemblages showed the opposite trend.

Table 3.2. Contingency tables for categorical variables which differed significantly from expected distributions of bird assemblages.

Rows: Understorey Columns: Bird assemblages[a]

	A1	A2-4	All	
open	18	4	22	Observed
	9.96	12.04	22.00	Expected
mod.	6	15	21	Observed
	9.51	11.49	21.00	Expected
dense	0	10	10	Observed
	4.53	5.47	10.00	Expected
All	24	29	53	
	24.00	29.00	53.00	

Chi-Square = 22.495, DF = 2, P-Value = 0.000
1 cells with expected counts less than 5.0

Rows: Surrounding landscape Columns: Bird assemblages[a]

	A1	A2-4	All	
rural	24	24	48	Observed
	21.74	26.26	48.00	Expected
urban	0	5	5	Observed
	2.26	2.74	5.00	Expected
All	24	29	53	
	24.00	29.00	53.00	

Chi-Square = 4.569, DF = 1, P-Value = 0.033
2 cells with expected counts less than 5.0

[a] For chi-squared analysis the higher level of division into two bird assemblages was used.

Table 3.3 Means and standard deviations of continuous variables and chi-squared analyses of ordinal variables for bird assemblages.

Variable	Assemblage 1 (n = 24[a])	Assemblage 2 (n = 13)	Assemblage 3 (n = 6)	Assemblage 4 (n = 10)	p[b]	Tukey's pairwise comparisons (indicates significantly different pairings)
Community-level variables						
Total species richness	14.042 ± 4.496	29.077 ± 3.861	31.0 ± 2.828	23.8 ± 4.826	**0.000**	A1-A2, A1-A3, A1-A4, A2-A3, A3-A4
Mean species richness	6.155 ± 1.446	12.993 ± 2.748	14.445 ± 2.083	9.212 ± 2.1	**0.000**	A1-A2, A1-A3, A1-A4, A2-A3, A3-A4
Local species richness	3.963 ± 0.724	5.769 ± 0.799	7.233 ± 0.437	5.66 ± 1.969	**0.000**	A1-A2, A1-A3, A1-A4, A2-A3, A3-A4
Species diversity	0.782 ± 0.127	1.136 ± 0.076	1.107 ± 0.059	1.079 ± 0.127	**0.000 (K-W)**	
Percentage dominance	59.9 ± 10.46	35.545 ± 5.74	36.572 ± 5.669	37.29 ± 12.16	**0.000**	A1-A2, A1-A3, A1-A4
Total bird density	6.621 ± 1.821	8.148 ± 2.854	10.272 ± 1.039	9.249 ± 4.758	**0.013**	A1-A3
Independent variables						
Area (ha)	19.6 ± 14.69	54.65 ± 32.78	101.85 ± 62.58	40.65 ± 31.98	**0.000**	A1-A2, A1-A3
Shape (perimeter/area)	126.11 ± 43.84	78.08 ± 27.48	87.82 ± 57.94	86.44 ± 49.91	**0.008**	A1-A2
Distance to 50 ha forest (m)	576.0 ± 805.6	588.5 ± 772.3	120.8 ± 103.0	1230.0 ± 1643.2	0.223	
Distance to 500 ha forest (m)	1533 ± 1518	2021 ± 1835	2004 ± 1513	2493 ± 2270	0.494	
Forest within 2.5 km (ha)	516.1 ± 238.2	445.3 ± 169.1	498.1 ± 353.2	530.0 ± 527.9	0.904	
Index of local patchiness	23.45 ± 15.59	27.72 ± 14.87	28.6 ± 25.53	38.71 ± 54.83	0.779	
Stems/hectare	345.6 ± 255.3	564.1 ± 305.6	658.6 ± 471.1	1186.0 ± 953.1	**0.002**	A1-A4
Stems <5cm dbh/ha	50.3 ± 75.8	181.8 ± 214.5	255.7 ± 250.2	674.5 ± 748.7	**0.010**	A1-A4
Stems 5-15cm dbh/ha	135.5 ± 148.6	191.2 ± 115.4	256.2 ± 223.7	336.0 ± 206.5	**0.018**	A1-A4
Stems 15-30cm dbh/ha	94.79 ± 70.02	102.87 ± 37.74	63.02 ± 33.11	107.62 ± 65.79	0.507	
Stems 30-60cm dbh/ha	41.81 ± 21.46	68.36 ± 37.55	51.52 ± 15.48	52.36 ± 18.8	0.073	
Stems >60cm dbh/ha	22.23 ± 13.22	19.34 ± 13.95	31.62 ± 10.93	15.51 ± 9.68	**0.033**	*
Index of vegetation structure	1.834 ± 2.883	0.428 ± 0.367	2.702 ± 4.989	0.177 ± 0.195	**0.028**	A1-A4
Non-eucalypts stems/ha	153.0 ± 155.3	206.8 ± 187.6	432.6 ± 442.7	935.0 ± 962.5	**0.001**	A1-A4, A2-A4
Eucalypt stems/non-eucalypt stems	6.92 ± 12.14	5.86 ± 14.26	2.83 ± 3.98	0.54 ± 0.51	0.155	
Stems ≤ 6 m/ha	121.8 ± 143.5	232.7 ± 233.5	400.6 ± 429.6	922.9 ± 920.9	**0.004**	A1-A4, A2-A4
Maximum eucalypt canopy height (m)	21.13 ± 5.311	21.417 ± 4.188	23.833 ± 3.371	18.0 ± 3.338	0.137	
Mean eucalypt canopy height (m)	12.357 ± 3.76	11.017 ± 2.9	13.45 ± 4.049	10.513 ± 2.671	0.310	
Median eucalypt canopy height (m)	12.091 ± 5.338	8.333 ± 4.583	14.6 ± 5.413	9.75 ± 2.659	0.088	

Mean eucalypt height as % of max. eucalypt height	59.6 ± 15.42	51.65 ± 10.48	55.37 ± 9.87	55.64 ± 7.69	0.377	
Basal area of trees (cm^2)/ha	234430± 69830	262300± 69300	310720± 46650	226940± 83670	0.085	
Basal area(cm^2)/no. of stems	1156.5 ± 975.7	607.2 ± 408.1	940.2 ± 1004.3	328.5 ± 281.6	**0.005**	A1-A4
Mean dieback score	3.7364 ± 0.690	3.2333 ± 0.816	2.62 ± 0.249	2.7125 ± 0.479	**0.001**	A1-A3, A1-A4
% eucalypts with epicormic shoots	39.28 ± 17.97	32.44 ± 15.84	15.34 ± 6.49	18.44 ± 8.21	**0.002**	A1-A3, A1-A4
% dead stems	20.53 ± 11.39	22.01 ± 9.35	15.75 ± 12.24	6.97 ± 4.9	**0.005**	A1-A4, A2-A4
Native plant species richness	58.3 ± 13.32	64.25 ± 14.23	55.4 ± 14.01	87.17 ± 20.54	**0.001**	A1-A4, A2-A4, A3-A4
Exotic plant species richness	22.217 ± 5.592	20.917 ± 6.127	23.6 ± 4.98	42.333 ± 10.95	**0.000**	A1-A4, A2-A4, A3-A4
Exotic plant species richness/native plant species richness	0.3965 ± 0.114	0.3267 ± 0.087	0.442 ± 0.112	0.4933 ± 0.108	**0.020**	A2-A4
Significance Index	19.96 ± 22.65	17.67 ± 17.82	12.8 ± 12.68	26.5 ± 22.13	0.781 (K-W)	
% trees logged	18.54 ± 14.13	7.33 ± 6.22	8.2 ± 9.23	0.6 ± 1.34	**0.005**	A1-A2, A1-A4
Mean annual precipitation (mm)	604.57 ± 37.74	586.17 ± 37.72	701.4 ± 66.23	662.17 ± 89.82	**0.000**	A1-A3, A2-A3, A2-A4
Precipitation in driest quarter (mm)	121.74 ± 8.71	120.58 ± 4.72	130.6 ± 15.32	138.67 ± 17.25	**0.011 (K-W)**	
Precipitation in wettest quarter (mm)	176.0 ± 13.15	169.75 ± 15.78	216.0 ± 26.8	192.5 ± 29.36	**0.000**	A1-A3, A2-A3
Mean annual temperature (°C)	11.004 ± 0.729	10.733 ± 0.713	11.32 ± 0.084	11.817 ± 0.392	**0.002 (K-W)**	
Altitude (m)	233.75 ± 118.2	274.62 ± 96.06	171.67 ± 37.1	128.0 ± 68.12	**0.005 (K-W)**	
Chi-squared[c]						
Overstorey recruitment					0.084	
Understorey density					**0.000**	
Geology					0.191	
Position in landscape					0.465	
Dominant eucalypt					0.821	
Floristic group					0.110	
Surrounding landscape					**0.033**	
Time since isolation					0.929	
Stocking rate					0.056	
Fire used as management tool					0.130	

[a] Not every variable was measured for every site, so the number of sites used in some calculations may have been fewer than the numbers in parentheses.

[b](K-W) indicates that a Kruskal-Wallis test was used to calculate differences in medians.

[c] Chi-squared analysis was performed using only the higher level of division into two bird assemblages. Contingency tables for those variables which significantly differed from normal are displayed in Table 3.2.

* indicates that the ANOVA was significant, but Tukey's pairwise comparison found no pairs of assemblages significantly different.

3.3.3 Community-level responses

Table 3.4 gives the significance levels on all relationships between community-level responses and independent variables.

Total species richness, mean species richness, Shannon-Wiener diversity index and dominance by the two most abundant species at a site showed similar trends (although the last of these in an opposite direction). All four showed very strongly significant relationships ($P \leq 0.001$) with remnant area and with understorey density. All four showed less strongly significant relationships with remnant shape, mean dieback scores and total stem density.

The following variables displayed significant relationships with some of these community-level responses: nature of surrounding vegetation; native species richness; exotic species richness; stems per hectare <5 cm dbh; stems per hectare 5-15 cm dbh; stems per hectare 30-60 ha dbh; index of vegetation structure; stems per hectare 6 m or shorter; basal area per hectare; basal area divided by stems; % trees logged; and eucalypts with epicormic shoots.

Local species richness showed slightly different relationships. It showed a strongly significant relationship ($P \leq 0.001$) with understorey. Less strongly significant relationships were found with the following variables: precipitation in wettest quarter; remnant area; total stems per hectare; stems per hectare 5-15 cm dbh; non-eucalypt stems per hectare; stems per hectare 6 m or shorter; basal area divided by stems; mean dieback score; and eucalypts with epicormic shoots.

Total bird density showed fewer significant relationships. These were with: mean annual precipitation; precipitation in the wettest quarter; understorey density; mean dieback score; eucalypts with epicormic shoots; and noisy miner abundance

3.3.4 Community-level responses in the absence of noisy miner colonies

The considerable differences between Assemblage One and the others confirm the merit of performing analyses excluding remnants supporting Assemblage One. Although noisy miners were present and abundant at many sites, remnants classified as supporting Assemblage One are defined by the present study as those dominated by noisy miner colonies. The results are presented in Table 3.5.

Far fewer significant relationships were evident when sites supporting noisy miner colonies were excluded from analysis. Total bird density showed no significant relationship with any independent variable. Landscape variables showed the most significant relationships. Remnant area correlated positively with total species richness and mean species richness, and negatively with dominance by the two most abundant bird species. Remnant shape correlated negatively with mean species richness and the Shannon-Wiener diversity index. Distance to the nearest forested area greater than 500 ha in size correlated negatively with total species richness and positively with dominance by the two most abundant bird species. Mean species number was higher in remnants located on laterite as opposed to other geological classes.

The only measure of vegetation structure that showed any significant relationship was density of stems 15-30 cm dbh, which correlated negatively with local species richness. Mean species number was significantly higher on sites with floristic group four (*E. amygdalina* dry forest) than on those with floristic group two (grassy *E. viminalis* woodland), while floristic group three was intermediate. Where noisy miner density was low, local species richness was significantly higher than at other sites.

These relationships make interesting comparisons with those found with the complete set of sites. Remnant area and shape show similar relationships, but with fewer community-level responses. One of the isolation measures displays some significant relationships, as do floristic and geological remnant characteristics, which was not the case when all sites were included. Independent variables, which displayed many significant relationships with the complete set of sites, such as tree health, noisy miner abundance, and vegetation structure, show almost no significant relationships with community-level responses.

Table 3.4. Significant relationships between independent variables and bird community-level responses.

(Correlations between continuous variables and ANOVA of community-level responses by categorical independent variables)

Community-level response	Total species richness[a]	Mean species richness	Local species richness	Species diversity[a]	% Dominance	Total bird density
Variable						
Mean annual temperature (°C)[a]	*	*	*	*	*	*
Mean annual precipitation (mm)[a]	*	*	*	*	*	0.05 (+)
Precipitation in driest quarter (mm)[a]	*	*	*	*	*	*
Precipitation in wettest quarter (mm)[a]	*	*	0.05 (+)	*	*	0.01 (+)
Altitude (m)[a]	*	*	*	*	*	*
Geology	*	*	*	*	*	*
Position in landscape	*	*	*	*	*	*
Area (ha)	0.001 (+)	0.001 (+)	0.05 (+)	0.001 (+)	0.001 (-)	*
Shape (perimeter/area)	0.001 (-)	0.001 (-)	*	0.001 (-)	0.01 (+)	*
Distance to 50 ha forest (m)	*	*	*	*	*	*
Distance to 500 ha forest (m)	*	*	*	*	*	*
Forest within 2.5 km (ha)	*	*	*	*	*	*
Index of local patchiness	*	*	*	*	*	*
Surrounding landscape	*	*	*	0.05	0.01	*
Time since isolation	*	*	*	*	*	*
Dominant eucalypt	*	*	*	*	*	*
Floristic group	*	*	*	*	*	*
Native plant species richness	*	*	*	*	0.05 (-)	*
Exotic plant species richness	*	*	*	*	0.05 (-)	*
Exotic plant species richness/native plant species richness	*	*	*	*	*	*
Significance index[a]	*	*	*	*	*	*
Stems/hectare	0.01 (+)	0.05 (+)	0.05 (+)	0.001 (+)	0.01 (-)	*
Stems <5cm dbh/ha	0.001 (+)	*	*	0.001 (+)	0.05 (-)	*
Stems 5-15cm dbh/ha	0.05 (+)	*	0.05 (+)	0.05 (+)	0.05 (-)	*
Stems 15-30cm dbh/ha	*	*	*	*	*	*
Stems 30-60cm dbh/ha	*	0.05 (+)	*	0.05 (+)	0.05 (-)	*
Stems >60cm dbh/ha	*	*	*	*	*	*
Index of vegetation structure	*	*	*	0.05 (-)	0.05 (+)	*

Non-eucalypts stems/ha	*	*	0.05 (+)	*	*	*
Eucalypt stems/non-eucalypt stems	*	*	*	*	*	*
Stems ≤ 6 m/ha	0.05 (+)	*	0.05 (+)	0.01 (+)	0.05 (-)	*
Maximum eucalypt canopy height (m)	*	*	*	*	*	*
Mean eucalypt canopy height (m)	*	*	*	*	*	*
Mean eucalypt height as % of max. eucalypt height	*	*	*	*	*	*
Basal area (cm^2)/hectare	*	0.05 (+)	*	*	*	*
Basal area/no. of stems	*	*	0.05 (-)	0.05 (-)	0.01 (+)	*
Overstorey recruitment	*	*	*	*	*	*
Understorey density	0.001	0.001	0.001	0.001	0.001	0.05
% trees logged	0.05 (-)	0.05 (-)	*	0.05 (-)	0.01 (+)	*
Mean dieback score	0.001 (-)	0.01 (-)	0.01 (-)	0.01 (-)	0.001 (+)	0.05 (-)
% eucalypts with epicormic shoots	0.05 (-)	*	0.01 (-)	0.05 (-)	0.01 (+)	0.05 (-)
% dead stems	*	*	*	*	*	*
Stocking rate	*	*	*	*	*	*
Fire used as management tool	*	*	*	*	*	*
Noisy miner abundance[a]	0.001 (-)	0.001 (-)	0.001 (-)	0.001 (+)	0.001 (-)	0.01 (-)
Common starling abundance[a]	*	*	*	*	0.05 (+)	*

(+) or (-) indicates the direction of correlation. Absence of this symbol indicates the relationship is established by ANOVA.

[a] These variables were correlated using Spearman's rank correlation coefficient.

* = not significant.

Table 3.5. Significant relationships between independent variables and bird community-level responses (excluding noisy miner colonies)

Community-level response	Total species richness[a]	Mean species richness	Local species richness	Species diversity[a]	% Dominance	Total bird density
Variable						
Mean annual temperature (°C)[a]	*	*	*	*	*	*
Mean annual precipitation (mm)[a]	*	*	*	*	*	*
Precipitation in driest quarter (mm)[a]	*	*	*	*	*	*
Precipitation in wettest quarter (mm)[a]	*	*	*	*	*	*
Altitude (m)[a]	*	*	*	*	*	*
Geology	*	*	*	*	*	*
Position in landscape	*	*	*	*	*	*
Area (ha)	0.001 (+)	0.001 (+)	*	*	0.05 (-)	*
Shape (perimeter/area)	*	0.05 (-)	*	0.05 (-)	*	*
Distance to 50 ha forest (m)	*	*	*	*	*	*
Distance to 500 ha forest (m)	0.05 (-)	*	*	*	0.01 (+)	*
Forest within 2.5 km (ha)	*	*	*	*	*	*
Index of local patchiness	*	*	*	*	*	*
Surrounding landscape	*	*	*	*	*	*
Time since isolation	*	*	*	*	*	*
Dominant eucalypt	*	*	*	*	*	*
Floristic group	*	*	0.05	*	*	*
Native plant species richness	*	*	*	*	*	*
Exotic plant species richness	*	*	*	*	*	*
Exotic plant species richness/native plant species richness	*	*	*	*	*	*
Significance index[a]	*	*	*	*	*	*
Stems/hectare	*	*	*	*	*	*
Stems <5cm dbh/ha	*	*	*	*	*	*
Stems 5-15cm dbh/ha	*	*	*	*	*	*
Stems 15-30cm dbh/ha	*	*	0.05 (-)	*	*	*
Stems 30-60cm dbh/ha	*	*	*	*	*	*

Stems >60cm dbh/ha	*	*	*	*	*	*
Index of vegetation structure	*	*	*	*	*	*
Non-eucalypts stems/ha	*	*	*	*	*	*
Eucalypt stems/non-eucalypt stems	*	*	*	*	*	*
Stems ≤ 6 m/ha	*	*	*	*	*	*
Maximum eucalypt canopy height (m)	*	*	*	*	*	*
Mean eucalypt canopy height (m)	*	*	*	*	*	*
Mean eucalypt height as % of max. eucalypt height	*	*	*	*	*	*
Basal area (cm^2)/hectare	*	*	*	*	*	*
Basal area/no. of stems	*	*	*	*	*	*
Understorey density	*	*	*	*	*	*
% trees logged	*	*	*	*	*	*
Mean dieback score	*	*	*	*	*	*
% eucalypts with epicormic shoots	*	*	*	*	*	*
% dead stems	*	*	*	*	*	*
Stocking rate	*	*	*	*	*	*
Fire used as management tool	*	*	*	*	*	*
Noisy miner abundance	*	*	0.01 (-)	0.05 (-)	*	*
Common starling abundance	*	*	*	*	*	0.05 (+)

(+) or (-) indicates the direction of correlation. Absence of this symbol indicates the relationship is established by ANOVA.

[a] These variables were correlated using Spearman's rank correlation coefficient.

* = not significant.

3.3.5 Species' distributions and abundances and independent variables

Table 3.6 displays those variables which differed significantly according to species' presence and abundance, both over the whole range of remnants and excluding remnants supporting noisy miner colonies. Significant relationships are presented in order of magnitude of significance for positiveness and negativeness. Adjustments to variables to be used for Chi-squared analysis excluding noisy miner colonies follow those described in the Methods. Even so, the smaller sample size made some Chi-squared tests invalid.

Analysis excluding noisy miner colonies used only presence-absence categories, *or* abundance classes. This is due to the fact that many species were present on almost all remnants that did not support noisy miner colonies, and the smaller number of "absent" sites invalidated presence-absence analysis. Symbols for significant relationships concerning abundance classes are more complex than for presence-absence. Significant results marked (++) indicate a broadly linear relationship between that habitat variable and species abundance. Those marked (+) indicate that the species is absent or rare where values for that variable are low, and that for higher values other variables determine the species' abundance. Those marked (+++) indicate that the species is present at high abundance at high values of the variable, and that for lower values other variables determine the species' abundance. Where cells are empty analysis was not done, and where no variables differed significantly this is noted.

Species are presented in five groups depending on their responses to competition from noisy miners and to habitat fragmentation (see discussion of this chapter and Appendix Two).

Table 3.6. Bird species and those variables which differ according to their distribution and abundance.

Species	All remnants	Excluding noisy miner colonies
Miner tolerant, unaffected by fragmentation		
noisy miner (abundance)	Mean dieback score*** (++), % eucalypts with epicormic shoots* (+), Remnant shape* (+++) Understorey density*** (---), Mean annual precipitation*** (-), Area** (---), Stems/ha* (---), Stems(5-15cm)/ha* (---), Trees <6m tall/ha* (---), Stems(5-15cm)/ha* (---) Mean canopy height/Max. canopy height* (?)	Mean dieback score* (+), % eucalypts with epicormic shoots* (+) Mean annual precipitation* (-)
grey butcherbird	Noisy miner abundance** (+), % dead stems** (+), Mean annual temperature* (+), Mean dieback score* (+), % eucalypts with epicormic shoots* (+) Non-eucalypt stems/ha* (-), Trees <6m tall/ha* (-),Understorey density* (-)	
grey butcherbird (abundance)	Noisy miner abundance*** (++), Understorey density*** (-), % eucalypts with epicormic shoots* (++), Area* (--)	Noisy miner abundance** (+), % dead stems* (+) Understorey density* (-)

Australian magpie	Mean dieback score*** (+), Basal area/no. of stems*** (+), % eucalypts with epicormic shoots** (+), % dead stems** (+), Noisy miner abundance** (+), Remnant shape* (+),Index of vegetation structure*** (+), Mean canopy height* (+) Understorey density*** (-), Stems/ha*** (-), Stems(<5cm)/ha*** (-), Stems(5-15cm)/ha*** (-), Non-eucalypt stems/ha*** (-), Trees <6m tall/ha*** (-)	
Australian magpie (abundance)	Noisy miner abundance*** (+++), Basal area/no. of stems*** (+), Mean dieback score***(++), % eucalypts with epicormic shoots** (++), Index of vegetation structure** (+), % dead stems* (++), Remnant shape* (++) Understorey density*** (--), Stems/ha*** (-), Stems(5-15cm)/ha*** (-),Trees <6m tall/ha*** (-), Stems(<5cm)/ha** (-), Non-eucalypt stems/ha** (-), Native plant species richness** (-), Exotic plant species richness* (-) Area*** (?), Area forested within 2.5 km* (?)	Noisy miner abundance** (+++), % dead stems* (++), Index of vegetation structure* (+), Basal area/no. of stems* (+) Stems/ha* (-), Stems(5-15cm)/ha* (-), Non-eucalypt stems/ha* (-), Trees <6m tall/ha* (-) Stems(15-30cm)/ha* (?)
eastern rosella	Noisy miner abundance*** (+), Basal area/no. of stems* (+), Mean dieback score* (+), % eucalypts with epicormic shoots* (+), % logging* (+) Understorey density** (-), Stems/ha* (-), Stems(5-15cm)/ha* (-), Non-eucalypt stems/ha* (-), Trees <6m tall/ha* (-)	Forest within 2.5 km* (-)

eastern rosella (abundance)	Noisy miner abundance*** (+++), Mean dieback score** (+++), % eucalypts with epicormic shoots** (++), Remnant shape* (++), Understorey density*** (---), Stems/ha* (--), Basal area/no. of stems* (++) Area* (?)	Noisy miner abundance*** (+++) Plant significance index* (?)
laughing kookaburra	Mean annual temperature** (+), Mean canopy height** (+), Basal area/ha** (+), Basal area/no. of stems* (+), Maximum canopy height* (+) Altitude* (-)	
laughing kookaburra (abundance)	Mean canopy height*** (+), Mean canopy height** (++), Basal area/ha** (++), Common starling abundance** (++), Index of vegetation structure* (+), % dead stems* (++), Basal area/no. of stems* (+) Stems(<5cm)/ha* (--) Stocking rate** (?), Mean annual temperature* (?), Floristic group** (*E. amygdalina* dry forest), Dominant eucalypt* (*E. amygdalina*),	Maximum canopy height** (++), Mean canopy height** (++), Index of vegetation structure** (++), % logging* (++), Basal area/no. of stems* (+) Stems(<5cm)/ha* (-), Stems(5-15cm)/ha* (--), Non-eucalypt stems/ha* (--), Trees <6m tall/ha* (--) Stocking rate** (?), Mean annual temperature* (?), *Use of fire** (no)
brown falcon	Stems(30-60cm)/ha*** (-), Mean canopy height/Max. canopy height*** (-), Noisy miner abundance* (-)	Stems(30-60cm)/ha*** (+), Distance to 50 ha forest* (+), Forest within 2.5 km** (-)
sulphur-crested cockatoo	% dead stems*** (+) Stems(<5cm)/ha* (-) *Floristic group*** (lowland *Themeda* grassy woodland), *Use of fire*** (no)	
sulphur crested cockatoo (abundance)	Forest within 2.5 km* (+) % dead stems*** (?), Mean dieback score* (?), Mean annual precipitation* (?), *Use of fire** (no)	% dead stems*** (+), Altitude* (?), *Dominant eucalypt species** (*E. amygdalina*), *Use of fire** (no)

forest raven (abundance)	% eucalypts with epicormic shoots* (+++)	Basal area/no. of stems** (++), Stock rate** (++), Index of vegetation structure** (++), % logging** (+), Eucalypt/non-eucalypt stems* (++) Non-eucalypt stems/ha** (--), Trees <6m tall/ha** (--), Stems/ha* (--), Stems(<5cm)/ha* (-), Stems(5-15cm)/ha* (--) *Use of fire*** (no)
striated pardalote (abundance)	Area*** (+), Understorey density*** (++) Remnant shape*** (-), Mean dieback score*** (-), Noisy miner abundance*** (-), % eucalypts with epicormic shoots** (-)	Area** (+), Altitude* (+) Mean dieback score** (-), % eucalypts with epicormic shoots* (-) Stems(5-15cm)/ha* (?), *Dominant eucalypt species*** (*E. amygdalina*)
welcome swallow	Index of local patchiness* (+), Use of fire* (no)	% dead stems* (+)
common bronzewing	none	Stems(>60cm)/ha* (+), Basal area/ha* (+)
common starling (abundance)	Remnant shape*** (++) Area** (---), Native plant species richness** (---)	Index of local patchiness** (--), Forest within 2.5 km* (--), Area* (---) Non-eucalypt stems/ha* (?), Eucalypt/non-eucalypt stems* (?)
Miner-tolerant, sensitive to fragmentation		
green rosella	Altitude* (+), Index of local patchiness* (+), % dead stems** (+), *Position*** (slope)	
green rosella (abundance)	Index of local patchiness* (++), Stems(30-60cm)/ha* (++) Distance to 50 ha forest** (--) Area** (?), Remnant shape* (?)	Area forested within 2.5 km* (+), % dead stems* (+) Distance to 50 ha forest** (-) *Use of fire** (no)
grey currawong	Forest within 2.5 km* (+), Index of local patchiness* (+), Stems/ha* (+), Trees <6m tall/ha* (+) Distance to 50 ha forest** (-)	Distance to 50 ha forest* (-)

Miner-intolerant, advantaged by fragmentation		
goldfinch	Forest within 2.5 km* (-), Index of local patchiness** (-), % logging** (-), *Dominant eucalypt species** (*E. viminalis*)	Remnant shape*** (+), Distance to 50 ha forest* (+), Common starling abundance* (+) Index of local patchiness*** (-), Forest within 2.5 km* (-), Area* (-), % logging* (-)
musk lorikeet	Stems/ha* (-), Stems(<5cm)/ha* (-), Basal area/no. of stems* (+), % logging* (+), Mean dieback score* (+)	not performed (insufficient occurrences)
blackbird	Mean annual temperature* (+), Mean annual precipitation** (+), Understorey density** (+), Distance to 50 ha forest* (+), Stems/ha* (+), Stems(<5cm)/ha* (+) Altitude*** (-), Forest within 2.5 km*** (-), Index of local patchiness*** (-), % logging** (-), Noisy miner abundance** (-), Stocking rate* (-), % eucalypts with epicormic shoots* (-) *Use of fire**** (yes)	Mean annual precipitation* (+), Common starling abundance* (+), Distance to 50 ha forest* (+) Forest within 2.5 km*** (-), Index of local patchiness*** (-), Altitude** (-), Remnant shape** (-), % dead stems* (-), % logging* (-) *Use of fire** (yes)
Miner-intolerant, unaffected by fragmentation		
yellow-rumped thornbill	Area** (+), Stems(30-60cm)/ha** (+), Understorey density** (+), Stems/ha* (+) Noisy miner abundance*** (-)	
yellow-rumped thornbill (abundance)	Area*** (?), Stems(30-60cm)/ha** (+), Remnant shape* (?), Understorey density* (?) Noisy miner abundance*** (-)	Mean dieback score* (++), Basal area/no. of stems* (+) Stems/ha* (--), Trees <6m tall/ha* (--), Understorey density* (-)

brown thornbill	Area*** (+), Understorey density*** (+), Distance to 500 ha forest* (+), Forest within 2.5 km* (+), Stems/ha* (+), Stems(<5cm)/ha* (+), Stems(5-15cm)/ha* (+), Stems(30-60cm)/ha* (+), Trees <6m tall/ha* (+) Noisy miner abundance*** (-), Mean dieback score*** (-), % eucalypts with epicormic shoots*** (-), Remnant shape** (-), Common starling abundance* (-)	
brown thornbill (abundance)	Understorey density*** (+), Area** (+) % logging* (?) Noisy miner abundance*** (-), Mean dieback score** (-), Noisy miner abundance*** (-), % eucalypts with epicormic shoots*** (-), Remnant shape* (-)	Noisy miner abundance* (--)
goshawk/sparrowhawk	Area*** (+), *Overstorey recruitment** (yes) Noisy miner abundance** (-), Remnant shape* (-), % logging* (-), Mean dieback score* (-)	none
dusky woodswallow	Area*** (+), Stems/ha** (+), Stems(<5cm)/ha** (+), Stems(5-15cm)/ha* (+), Understorey density* (+), Noisy miner abundance*** (-), Common starling abundance* (-) Basal area/no. of stems* (-), % logging* (-), Mean dieback score* (-),Remnant shape* (-)	
dusky woodswallow (abundance)	Area** (+), Stems/ha* (+), Understorey density* (+) Common starling abundance* (?) Noisy miner abundance*** (-), % logging* (-)	none

fantailed cuckoo	Area*** (+), Basal area/ha* (+), Understorey density* (+) Noisy miner abundance*** (-), Remnant shape* (-), Mean dieback score* (-), Stocking rate* (-)	none
shining bronze-cuckoo	Area** (+), Understorey density** (+) Noisy miner abundance** (-)	none
spotted pardalote	Area*** (+), Stems/ha*** (+),Understorey density** (+), Stems(5-15cm)/ha** (+), Stems(<5cm)/ha* (+),Trees <6m tall/ha* (+) Basal area/no. of stems*** (-),Remnant shape*** (-), Noisy miner abundance*** (-), Stems(>60cm)/ha*** (-), Mean canopy height*** (-), Index of vegetation structure** (-), Maximum canopy height** (-), Mean canopy height/Max. canopy height* (-), % logging* (-), Mean dieback score* (-)	
spotted pardalote (abundance)	Stems/ha*** (+), Stems(<5cm)/ha*** (+), Stems(5-15cm)/ha*** (+), Trees <6m tall/ha*** (+), Understorey density** (+), Non-eucalypt stems/ha** (++) Noisy miner abundance*** (-), Mean canopy height/Max. canopy height*** (-), Mean dieback score*** (-), Basal area/no. of stems*** (-), Index of vegetation structure** (-), Mean canopy height** (-), % eucalypts with epicormic shoots* (-) Area*** (?), Remnant shape*** (?), *Dominant eucalypt species*** (*E. amygdalina*),	Stems/ha* (++), Stems(<5cm)/ha* (++), Stems(5-15cm)/ha* (++), Non-eucalypt stems/ha* (+), Trees <6m tall/ha* (++), Mean canopy height/Max. canopy height** (-), Index of vegetation structure* (-), Mean canopy height* (--), Basal area/no. of stems* (--)
flame robin	Area* (+),Understorey density* (+) Noisy miner abundance*** (-)Mean dieback score** (-), Remnant shape* (-)	Stems(30-60cm)/ha* (+) Noisy miner abundance** (-)

grey fantail	Area*** (+), Understorey density*** (+), Stems/ha* (+), Stems(30-60cm)/ha* (+), Trees <6m tall/ha* (+), Noisy miner abundance*** (-)Remnant shape*** (-), Mean dieback score*** (-), % eucalypts with epicormic shoots** (-), Common starling abundance* (-)	
grey fantail (abundance)	Mean annual precipitation*** (+++), Understorey density** (+), Area* (++) Noisy miner abundance*** (--), Mean dieback score** (--) Stems(30-60cm)/ha* (?)	Mean canopy height/Max. canopy height** (+), Mean canopy height* (++), Mean annual precipitation* (++) Noisy miner abundance** (---), Stems(15-30cm)/ha* (-)
silvereye	Understorey density*** (+), Area** (+), Stems/ha** (+), Non-eucalypt stems/ha** (+), Trees <6m tall/ha* (+), Stems(5-15cm)/ha* (+) Noisy miner abundance*** (-), % logging** (-), Remnant shape** (-), Mean dieback score** (-), % eucalypts with epicormic shoots** (-), Index of vegetation structure* (-), Basal area/no. of stems* (-), Stocking rate* (-)	Noisy miner abundance* (-), Stems(15-30cm)/ha* (-)
superb fairy-wren	Understorey density** (+) Noisy miner abundance*** (-), Mean dieback score** (-), % eucalypts with epicormic shoots** (-),	
superb fairy-wren (abundance)	Understorey density ** (++), Mean annual precipitation* (++), Noisy miner abundance*** (--), Mean dieback score* (-), % eucalypts with epicormic shoots* (-),	Stems(15-30cm)/ha* (--)

Miner-intolerant, sensitive to fragmentation		
yellow wattlebird	Area*** (+), Stems/ha*** (+), Stems(5-15cm)/ha*** (+),Trees <6m tall/ha*** (+), Understorey density*** (+), Stems(<5cm)/ha** (+), Index of vegetation structure** (-), Non-eucalypt stems/ha** (+), Native plant species richness** (+), Exotic plant species richness(+) Remnant shape*** (-), Basal area/no. of stems*** (-), Noisy miner abundance*** (-), % logging*** (-), Mean dieback score** (-), % eucalypts with epicormic shoots* (-), Common starling abundance* (-), Stocking rate* (-)	
yellow wattlebird (abundance)	Area*** (+), Understorey density*** (+), Stems/ha** (+), Stems(5-15cm)/ha** (+), Native plant species richness** (+), Exotic plant species richness** (++), Stems(<5cm)/ha* (+), Trees <6m tall/ha* (+), Non-eucalypt stems/ha* (+) Noisy miner abundance*** (-), % logging*** (-), Basal area/no. of stems** (--), Common starling abundance* (--), Index of vegetation structure* (-), Mean dieback score* (--), % eucalypts with epicormic shoots* (-), Remnant shape* (-) *Use of fire** (yes)	Exotic plant species richness** (++), Native plant species richness* (+) Distance to 500 ha forest** (-) *Use of fire** (yes)
grey shrike-thrush	Area*** (+), Understorey density*** (+), Stems/ha** (+), Stems(5-15cm)/ha* (+), Stems(30-60cm)/ha* (+), Trees <6m tall/ha* (+) Noisy miner abundance*** (-), Remnant shape*** (-),Mean dieback score*** (-), % eucalypts with epicormic shoots** (-), % logging** (-), Basal area/no. of stems* (-)	

grey shrike-thrush (abundance)	Forest within 2.5 km** (+++), Index of local patchiness** (+++), Area*** (+), Understorey density*** (+) Distance to 50 ha forest** (---), Noisy miner abundance*** (-), Remnant shape** (-), % eucalypts with epicormic shoots** (--), Common starling abundance** (--) Stems/ha* (?), Stems(<5cm)/ha* (?), Stems(5-15cm)/ha* (?), Stems(30-60cm)/ha* (?), Trees <6m tall/ha* (?), % logging* (?)	Forest within 2.5 km*** (++), Index of local patchiness** (++) Distance to 50 ha forest*** (---),
black-faced cuckoo-shrike	Area*** (+), Understorey density* (+) Noisy miner abundance*** (-), Forest within 2.5 km** (-), Index of local patchiness* (-), Remnant shape* (-)	
black-faced cuckoo-shrike (abundance)	Area*** (+), Understorey density** (+), Stems/ha** (++), Stems(<5cm)/ha* (+), Stems(5-15cm)/ha* (++), Trees <6m tall/ha* (++) Noisy miner abundance*** (-), Mean dieback score** (-), Basal area/no. of stems* (--) *Use of fire** (yes), Remnant shape* (?)	Area* (+) *Use of fire*** (yes)
pallid cuckoo	Area*** (+), Stems/ha** (+), Stems(<5cm)/ha** (+), Understorey density** (+), Trees <6m tall/ha* (+) Noisy miner abundance*** (-), Remnant shape** (-), % logging** (-), Basal area/no. of stems* (-), Mean dieback score** (-), Index of vegetation structure* (-), Common starling abundance* (-) Position* (hilltop), *Floristic group** (grassy *E. viminalis* woodland), Use of fire* (yes),	Area** (+)

yellow-throated honeyeater	Area*** (+), Stems/ha*** (+), Understorey density*** (+), Stems(<5cm)/ha** (+), Stems(5-15cm)/ha** (+),Non-eucalypt stems/ha** (+), Trees <6m tall/ha** (+) Noisy miner abundance*** (-), Remnant shape*** (-), Mean dieback score*** (-), % eucalypts with epicormic shoots*** (-), Basal area/no. of stems** (-), % logging** (-), Stocking rate* (-), Index of vegetation structure* (-)	
yellow-throated honeyeater (abundance)	Trees <6m tall/ha*** (+), Understorey density*** (+), Stems/ha*** (+), Stems(<5cm)/ha*** (+), Stems(5-15cm)/ha*** (+), Non-eucalypt stems/ha** (+), Area** (+) Noisy miner abundance*** (-), Index of vegetation structure*** (-), Basal area/no. of stems*** (-), Mean dieback score** (-), % logging** (--), Mean canopy height* (--) Remnant shape** (?)	Stems/ha** (+), Stems(<5cm)/ha** (+), Stems(5-15cm)/ha** (+), Trees <6m tall/ha* (+) Basal area/no. of stems*** (-), Mean canopy height*** (-), Index of vegetation structure** (-), Maximum canopy height** (-), Distance to 500 ha forest** (-), Mean canopy height/Max. canopy height* (--) Stocking rate* (?)
dusky robin	Area*** (+), Understorey density* (+), Index of local patchiness* (+) Remnant shape*** (-), Noisy miner abundance** (-), Distance to 50 ha forest** (-), Forest within 2.5 km* (-), Common starling abundance* (-)	Area** (+), Forest within 2.5 km** (+), Index of local patchiness** (+), % logging* (+) Distance to 50 ha forest** (-), Remnant shape* (-) *Dominant eucalypt species** (*E. amygdalina*)
golden whistler	Area*** (+), Stems/ha** (+), Stems(<5cm)/ha** (+), Stems(30-60cm)/ha** (+), Understorey density** (+), Stems(5-15cm)/ha* (+), Trees <6m tall/ha* Noisy miner abundance*** (-), Remnant shape** (-), Mean dieback score** (-), % logging* (-), (+), Basal area/no. of stems* (-)	Area* (+), Basal area/ha* (+) Noisy miner abundance* (-), Distance to 50 ha forest* (-)

scarlet robin	Area*** (+), Stems/ha*** (+), Understorey density*** (+), Stems(<5cm)/ha** (+), Stems(5-15cm)/ha** (+), Trees <6m tall/ha** (+), Stems(30-60cm)/ha* (+), Non-eucalypt stems/ha* (+) Mean dieback score*** (-), Noisy miner abundance*** (-), Common starling abundance** (-), % eucalypts with epicormic shoots** (-), Remnant shape** (-), Basal area/no. of stems** (-), % logging** (-), Index of vegetation structure* (-), Stocking rate* (-)	
scarlet robin (abundance)	Area*** (+), Understorey density** (+) Noisy miner abundance*** (-), Remnant shape* (-), Distance to 50 ha forest* (--) Stems/ha* (?), Stems(5-15cm)/ha* (?), Trees <6m tall/ha* (?), Mean dieback score* (?)	Distance to 50 ha forest* (--) Maximum canopy height* (?)
crescent honeyeater	Area*** (+), Understorey density*** (+), Mean annual temperature* (+), Stems/ha* (+), Stems(5-15cm)/ha* (+), Trees <6m tall/ha* (+) Mean dieback score*** (-), % eucalypts with epicormic shoots*** (-), Noisy miner abundance*** (-), Remnant shape** (-), Basal area/no. of stems* (-)	Area forested within 2.5 km* (+), Basal area/ha* (-), % eucalypts with epicormic shoots* (-)

crescent honeyeater (abundance)	Area*** (+), Understorey density*** (+), Native plant species richness*** (++), Exotic plant species richness** (++), Trees <6m tall/ha** (+++), Mean annual temperature* (+++), Stems/ha* (++), Stems(<5cm)/ha* (+++), Stems(5-15cm)/ha* (++), Non-eucalypt stems/ha* (+++), Mean annual precipitation* (+++) Noisy miner abundance*** (--), Mean dieback score** (--), % eucalypts with epicormic shoots** (--), Remnant shape** (--), Basal area/no. of stems* (---), % dead stems* (---), % logging* (-), Stocking rate* (---), Index of vegetation structure* (---) Use of fire** (yes)	Native plant species richness* (++), Exotic plant species richness* (++) % dead stems* (---) Use of fire* (yes)

*indicates $P \leq 0.05$, ** indicates $P \leq 0.01$, *** indicates $P \leq 0.001$

(-) or (+) indicates that the variable was lower or higher where the species was present

For species abundance classes (+) indicates that values for classes 2 & 3 > those for class 1, (++) indicates that values for class 3 > those for class 2 > those for class 3, and (+++) indicates that values for class 3 > those for classes 1 & 2. (-), (--), and (---) indicate the opposite trends, and (?) indicates no linear pattern.

Underlined indicates that the relationship was determined using a Kruskal-Wallis test.

Italics indicates a chi-squared test with one or more expected value < 5.

Table 3.6 presents a complex picture of the relationships of species with habitat variables, and it must be emphasised that the variables tested do not approach an exhaustive list. Each species shows its own relationships, and the used of abundance classes and of a data set excluding noisy miner colonies further informs those relationships. Some trends are apparent. Most of those species which do not occur in Assemblage One show strong relationships (either positive or negative) with those variables which discriminate Assemblage One from the other assemblages: remnant area, stem density and tree health. Species that were commonly found in Assemblage One show the opposite trends. However, when these remnants are excluded from the analysis, many of these relationships disappear or lessen in significance. In some cases they have been reversed. For example, the yellow-rumped thornbill shows a positive relationship with understorey density over all remnants, but in the reduced subset it shows a negative relationship. Other species generally considered to prefer open country but which show positive relationships with stem density include the dusky woodswallow, black-faced cuckoo-shrike and superb fairy-wren. Again, these are artefacts of their absence from the open woodland dominated by noisy miner colonies, and these relationships are not apparent when miner colonies are excluded.

Some species show no significant relationships either over all remnants or for the reduced subset, but no species shows no significant relationships for both. The four species that show no significant relationships for the reduced subset (goshawk/sparrowhawk, shining bronze-cuckoo, fantailed cuckoo and dusky woodswallow) are intolerant of noisy miner colonies, but otherwise appear randomly likely to be present in a remnant, at least in terms of the habitat variables measured.

Noisy miner abundance was a strong predictor of the presence and/or abundance of many species, although that predictive ability reduced considerably in the absence of miner colonies. In fact, noisy miner abundance was a significant variable for twenty-two species for all remnants, but that reduces to four species when miner colonies are excluded. Common starling abundance proved significant for several species, but fewer than might have been expected considering their abundance throughout the study, and not for tree hollow-nesting species, which might have been expected to suffer most from their presence. Some hollow-nesting species (green rosella, sulphur-crested

cockatoo, laughing kookaburra) were present where the number of dead stems was high.

Isolation measures proved to be important for several species. Notable amongst these were the grey shrike-thrush, dusky robin and grey currawong. It is worth noting that the isolation measure that proved significant for species richness was distance to the nearest forest of at least 500 ha, while for individual species the significant measure was distance to nearest forest of at least 50 ha.

Some measures of climate showed significant relationships. Of these, the most notable may be that noisy miners were abundant where precipitation was low. Other relationships worth mentioning include the negative relationships with canopy height shown by the spotted pardalote and yellow-throated honeyeater, and the positive relationships shown by the grey fantail and laughing kookaburra. Use of fire as a management tool also proved to be important for several species. Those which showed preference for use of fire tended to be those which were advantaged by denser stem density (yellow wattlebird, pallid cuckoo, crescent honeyeater, blackbird), while those which were more common where fire was not used as a management tool tended to be those which prefer open country (sulphur-crested cockatoo, welcome swallow, green rosella, forest raven).

3.4 Discussion

3.4.1 Species observed

The suite of bird species recorded is normal for dry sclerophyll forest and eucalypt woodland in Tasmania. Bosworth (1976) recorded 50 bird species from eight remnants. Only two species, the brown quail (*Coturnix ypsilophora*) and the beautiful firetail (*Stagonopleura bella*) were recorded in that study and not in this one, and both of these are more typical of wetter habitats and heathland. Other studies of smaller areas over time have recorded between 43 and 55 species (Ratkowsky 1983, Thomas 1986, Dickinson *et al.* 1986). These studies have also recorded a handful of birds not recorded in this study,

but again most of them are typical of wetter habitats. Of dry eucalypt forest and woodland species, only the satin flycatcher (*Myiagra cyanoleuca*) and the spotted quail-thrush (*Cinclosoma punctatum*) might have been expected to be recorded and were not. Five species endemic to Tasmania (yellow wattlebird, yellow-throated honeyeater, dusky robin, grey currawong and green rosella) were commonly found in the study area, as was one species for which Tasmania is a stronghold of distribution (crescent honeyeater). The swift parrot, a breeding endemic, was recorded from a small number of remnants.

The common starling was the only species recorded from all sites, although it was not abundant everywhere. In Autumn this species formed large flocks, and in Spring it nested intensively at several sites, with constant movement to and from nests. The noisy miner was the second most abundant species recorded, after the common starling, and it was totally absent only from four urban or semi-urban sites. This species was frequently observed vigorously chasing and mobbing other birds, confirming the experience elsewhere in Australia (Dow 1977, Grey *et al.* 1998). Other widespread species included the forest raven, Australian magpie, grey butcherbird, and striated pardalote, all species which were either impervious to miner attacks or which were able to evade them. Species which were unable to tolerate attacks by noisy miners were less widespread, but were often abundant in the absence of noisy miner colonies. Chief amongst these were the brown thornbill and grey fantail, but other species which were present (and frequently abundant) in most remnants in the absence of noisy miner colonies included the grey shrike-thrush, crescent honeyeater, scarlet robin, yellow-throated honeyeater and spotted pardalote. The role of the noisy miner in influencing distributions of other bird species and in confusing the interpretation of species' habitat preferences by the vigorous exclusion of many species from remnants is discussed in more detail below.

No species was restricted to the control transects, and most species recorded were moderately widespread, at least in the absence of noisy miner colonies. Species restricted in distribution over this study generally fell into one of four categories. Firstly, water birds, whose presence was dependent on wetland habitat either in or very near the remnant. These included the chestnut teal, white-faced heron and Australian shelduck. Secondly, species generally restricted to open, agricultural country which were recorded from remnants into which they had "strayed". These included the Tasmanian native-

hen, Richard's pipit and masked lapwing. Thirdly, wide-ranging species which are probably uncommon over the entire landscape but which were recorded from a small number of remnants as a result of their movements. These included the peregrine falcon, wedge-tailed eagle, and yellow-tailed black cockatoo. Fourthly, species which have a urban bias, whether due to introduction, such as the house sparrow and galah, or due to exploitation of a new source of food, such as the New Holland honeyeater.

3.4.2 Bird assemblages

Experience in the field combined with classification and ordination demonstrated a striking difference between Assemblage One and the other assemblages. Rather than a continuum, as may have been expected if species were responding individually to habitat gradients, there is a radical change which is clearly apparent in the field. This clear change is contrary to the experience of Recher *et al.* (1991) in southeastern New South Wales, where the avifauna did not form discrete communities. Noisy miners were not recorded in that study: bell miners were but are not mentioned as a dominant element of the avifauna. The results of their study correspond more closely with the differences between the other three assemblages, which are considerably more subtle. The commoner dry sclerophyll forest and eucalypt woodland species are generally present on sites supporting all three assemblages but at differing frequencies and abundances, suggesting that responses to habitat variables and resource availability by individual species are responsible for the division of these sites into different assemblages.

At least one of the six control transects was classified into each of the four bird assemblages. Combined with the fact that no bird species was restricted to the control transects, this indicates that the avifauna of large continuous areas of forest (ranging from 590-7000 ha) is not different from that of the smaller remnants as a whole. Assemblage One was present in a control transect which was located at the edge of a large area and which had a very open understorey. Avifaunal differences between edges and interiors of remnants are discussed further in Chapter Six. However, the fact that the other five control transects supported one of the other three assemblages suggests

that the proportion of forested area which supports Assemblage One is greater in highly fragmented landscapes..

Assemblage Four was largely found in urban or semi-urban situations, and several species probably frequented nearby gardens as well as the remnant or even used the remnant only incidentally as part of a matrix which is artificially very floristically diverse. This floristic diversity is reflected in the fact that Assemblage Four occurred on sites significantly higher in both exotic and native plant species richness. The presence of the swift parrot in three of these remnants reflected the presence of its preferred food species, *Eucalyptus globulus*, either on the site itself or very close by. The absence of open country species, such as the Australian magpie and eastern rosella, from many of these sites may be explained by the lack of feeding opportunities in the urban landscape surrounding them. The former species is quite common in suburban areas on mainland Australia, but much less so in Tasmania (personal observation). The latter is common in some urban situations in Tasmania, but is probably disadvantaged by the density of the understorey and lack of grass cover in the remnants supporting Assemblage Four. The independent variables which distinguished Assemblage Four were largely concerned with stem density, and these can also mostly be explained by the location of the sites in urban areas. These sites generally had large numbers of small trees regenerating, with a high proportion of non-eucalypts (*Allocasuarina verticillata, Beyeria viscosa* and *Acacia mearnsii*), and low levels of logging, all of which reflect the opportunity for regeneration where human activity is less interventionist. Tree health was also relatively good, with the low proportion of dead stems again reflecting a high density of small live stems and eucalypt dieback scores reflecting a large number of small eucalypts, which are more likely to be in good health.

Assemblages Two and Three were the most similar pairing, and the dividing line between them was blurred, although Assemblage Three was more species rich. Assemblage Three tended to have fewer open country and edge species, which possibly reflects the fact that it was found in the largest remnants. These two assemblages contained most of the species which are commonly associated with dry sclerophyll forest and eucalypt woodland in Tasmania, including several species of honeyeater, two species of thornbill and three species of robin. All of these species were noticeably absent from noisy miner colonies. For both assemblages, stem density in the smaller

dbh categories was intermediate, while Assemblage Three had the highest values for the largest trees. Tree health and logging proportions also differentiated remnants with these two assemblages from those with Assemblage One.

In contrast with the similarities displayed between the other three assemblages, the major differences between Assemblage One and the others suggests a threshold or thresholds which have altered the direction of species composition on these sites. Remnants classified as supporting Assemblage One were effectively those which were entirely dominated by noisy miner colonies, and the presence of these colonies had a remarkable impact on the avifauna of the remnants. Assemblage One was markedly less rich or diverse in bird species, averaging less than half as many species than Assemblages Two and Three. The two most abundant species (noisy miner and either common starling or eastern rosella) averaged almost 60% of all birds at a site, almost twice as much as for other assemblages. Of the species commonly recorded in Assemblage One only the striated pardalote was not a large bird characteristic of open country and edge habitats. This assemblage is very similar to that recorded in dieback-affected eucalypt woodland in New England by Ford & Bell (1981). The preponderance of large birds may explain the lower total bird density as compared with the other assemblages, rather than a lack of resources. The large size of these birds also explains their resistance to noisy miner aggression. The striated pardalote nests in small tree hollows, which may provide it with refuge from noisy miner attacks, which were observed quite frequently. It seems to be the only small bird which is able to persist in areas where noisy miners are abundant, although individuals were observed being chased quite often. Recher (pers. comm.) noted that the striated pardalote nests in noisy miner colonies on the New England tablelands of New South Wales, but that it does not feed within the colonies and thus avoids conflict.

The behaviour of noisy miners and bell miners and the establishment of miner colonies have been well documented. New miner colonies are more likely to arise from small groups leaving established colonies, rather than the movements of individual birds or pairs, and it may be that a minimum number of birds is needed to defend the nascent colony (Clarke & Fitzgerald 1994). Noisy miners were originally thought to breed communally (Dow 1979), but DNA profiling has shown that they are monogamous rather than cooperatively polyandrous (Poldmaa *et al.* 1995). Females are highly

territorial within the colony, and up to twenty males may help the breeding pair. These trends are also apparent in the bell miner (Clarke 1988, Clarke & Fitzgerald 1994). Within the colony, bell miners form a loose aggregation of discrete and mobile groups based on the family unit (Smith & Robertson 1978). These groups hold separate territories but as a colony they cooperate to mob intruders. Colonies of both species are aggressively policed throughout the year, and almost all other bird species are excluded by miners acting in concert to chase away and even kill other birds. These attacks do not seem to be related to the threat presented by the intruders, either as predators or as competitors, and even reptiles and mammals may be mobbed (Poiani *et al.* 1990, Dow 1977).

These attacks are sufficient to exclude most other bird species from miner colonies and to radically affect both the number and the type of species that remain. Poiani *et al.* (1990) found that the decrease of honeyeaters in a park in Victoria was correlated with the local increase of bell miners over a period of seven years. Several other studies have noted a relationship between the presence of either species of miner and bird species richness (Low 1994, Er 1997). Loyn (1987) called noisy miners a special factor in patches of Victorian eucalypt forest, noting that few other birds could nest where they were abundant. Noisy miner abundance improved species richness regression relationships when included as a patch character. The evidence for interspecific competition by miners as a depressing influence on species richness is supported by a series of experiments (Loyn *et al.* 1983, Grey *et al.* 1998) in which miners were removed from patches of eucalypt forest. Following removal, there was an influx of small insectivorous birds. Miners did not reinvade, even when colonies were nearby, which supports the theory that they establish themselves only in groups.

Assemblage One was generally found on the smallest remnants, and also on those remnants with the lowest values for total stem density (particularly in the smallest dbh classes), stem density for non-eucalypts and stem density for short trees. In addition, sites supporting Assemblage One had the highest rates of logging and the highest levels of dieback and tree death. It is suggested that these relationships present a unified story regarding the distribution of bird assemblages, bird species richness and species' distributions.

The strong relationship between remnant area and assemblage distribution suggests the presence of a size threshold of around 20-30 ha, below which noisy miner colonies were likely to dominate the entire remnant, and above which one of the other three assemblages would be expected to be present. The presence of dense understorey in small remnants or of open understorey in large remnants could alter these outcomes. Assemblage One was strongly associated with low stem density and with high logging rates. Where the understorey is more open, noisy miners chase away almost all birds and feed in all zones rather than being mainly restricted to foliage (Dow 1977). Dow also noted that noisy miner colonies can number up to several hundred birds and occupy 40 ha, which means that a single colony can completely dominate smaller isolated woodlands. It seems that this may be the case in this study, but that where understorey is sufficiently dense competitive behaviour by noisy miners is less effective and other species can coexist, forming a richer and more diverse avifauna. Understorey regeneration has been suggested as having potential for mitigating the dominance of noisy miners elsewhere in Australia (Grey *et al.* 1998). However, it is important to note that remnant area was not correlated with measures of stem density, apart from a positive relationship with stems 30-60 cm dbh. Small remnants may be considered in poorer condition in some respects, but the measures of vegetation structure did not differ significantly. This suggests that remnant area and stem density act independently to influence the distribution of noisy miner colonies.

The presence of a remnant size threshold at around 20-30 ha, below which most remnants supported a depauperate avifauna, supports the findings of Loyn (1987) in Victorian forests, and of Catterall *et al.* (1997) in south-east Queensland, although both of these studies placed the size threshold at around 10 ha. Both of these studies also emphasised the role of noisy miners in affecting bird diversity and noted that miner colonies tended to be found in small remnants that were in poor condition. The idea of a size threshold which also relies on vegetation structure concurs with comments by Lynch & Whigham (1984), who stated that "above some critical minimum patch area, floristically and physiognomically 'rich' forests may support bird communities that are quantitatively and qualitatively similar to those found in larger, but 'poorer' forests." Dominant eucalypt species showed no relationship with assemblage distribution, but

vegetation physiognomy can apparently play a part, as it allows other assemblages to occupy remnants which are of a size where Assemblage One might be expected.

The other set of variables that differentiated bird assemblages was climatic, and the reasons for this are less clear. Assemblage Three occurred on the warmest and wettest sites, and this may help explain its differentiation from Assemblage Two, although the reasons for this are not apparent. Higher precipitation probably results in denser vegetation, which is less amenable to the establishment of noisy miner colonies, but the mechanisms by which climate influences the distribution of bird assemblages are not apparent. However, the strong relationships between the distribution of Assemblage One, measures of tree health and precipitation are discussed in greater detail in Chapter Four.

The division of the avifauna into bird assemblages reflects a combination of influences: vegetation structure, landscape features, and competition. The overwhelming influence on the birds is the presence of noisy miner colonies and the aggressive behaviour of this species. However, these colonies are only present where habitat is appropriate, in small, open and disturbed remnants. Elsewhere, Assemblages Two and Three probably represent something approximating a natural avifauna, and Assemblage Four probably represents the influence of another form of disturbance: urban expansion. In the agricultural landscape of the Midlands, this suggests that small remnants in poor condition are likely to support noisy miner colonies, which restrict the presence of many small insectivorous bird species. However, where the undergrowth and understorey of such remnants is relatively dense, presence of miner colonies is not a foregone conclusion, as this density inhibits the colonisation of forest by miners.

3.4.3 Bird community-level responses

The trends in bird community-level responses are very much the same trends seen in bird assemblage distribution. This is not surprising, considering the active exclusion of many species which is practised by miners, and the strong relationship between miner colonies and vegetation structure, tree health and remnant area. Similar patterns of reduced species diversity and richness have also been observed on mainland Australia in

noisy miner and bell miner colonies (Loyn 1987, Grey *et al.* 1998, Clarke & Schedvin 1999). Exclusion of noisy miner colonies from the analysis presents a very different picture, one which is more in keeping with studies elsewhere in the world. While it must be recognised that this eliminates what may be considered a valid subset of the remnants from consideration, those which were excluded tended to be the most disturbed and in the least natural condition.

Lower total bird density where noisy miners are more abundant probably reflects the larger size of the birds that can coexist with that species rather than a decrease in the productivity of those remnants. Conversion of bird numbers into biomass may well have produced no significant result. Indeed, due to the fact that many of the species present in noisy miner colonies feed partly or predominantly in the surrounding landscape, one might expect biomass to be higher. The other independent variables which were correlated significantly with total bird density also relate to the presence of noisy miner colonies. None of these trends were evident in the reduced subset, indicating that these forests and woodlands tend to be comparable in terms of resource production for birds. In England, Ford (1987) found that total bird density was greater in small woods, and attributed this to increased use by birds of surrounding countryside.

The measures of bird species richness, diversity and dominance showed strong relationships with a wider range of independent variables, particularly area, understorey openness, stem density, noisy miner abundance and tree health. The interconnected nature of the influences of all of these variables make it difficult to be certain how much of the decrease in species richness and diversity in remnants dominated by noisy miners is due to interspecific competition and how much is due to changes in habitat which affect species richness. The most important influence on these community-level responses appears to be interspecific competition arising from the presence of noisy miner colonies (with additional pressure from associated species such as the grey butcherbird in the form of predation). However, the distribution of miner colonies is determined by remnant area and vegetation structure, and presence of a relatively dense understorey reduces the efficacy of noisy miner attacks (Dow 1977), preventing the exclusion of some species. Nevertheless, it might be expected that vegetation structure and area would influence bird species richness and diversity (positively), and dominance by abundant species (negatively) independently of the relationship of miner colonies

with these variables. This would be due to the existence of more niches where an understorey was present, or due to greater habitat diversity that may exist in larger remnants. The fact that vegetation structure did not show significant relationships with community-level responses in the absence of miner colonies suggests that it does not have this independent role. Area did maintain some of those significant relationships, albeit less strong ones, which suggests that it does. Most of the larger remnants in the present study supported a relatively diverse avifauna, although where the understorey was open and noisy miner colonies were present, measures of species richness and diversity approximated those of small remnants.

It must be remembered that analysis of remnants excluding noisy miner colonies removed most open and disturbed remnants from analysis, and that in the absence of noisy miner colonies remnants of this type may well still maintain a relatively small number of species. In the absence of noisy miner colonies, area and its related variable, remnant shape, emerged as the most important factors predicting species richness, although the relationships were not with all community-level responses. The increase in species richness with remnant area is probably due to an increase in the number of habitat types present. This probably does not relate to the presence of different interior forest habitats, nor even the presence of such elements as wetlands, as these were largely avoided in selecting sites. Rather, it most probably refers to the presence of edge habitats in addition to those of forest interior. Most of the species, which would be considered to show affinity for open country, were recorded from a majority of remnants, although their abundances were variable. As a result larger remnants tend to have many of the species which are present in small remnants, but also have species which preferred forest interiors.

Increased total species richness as remnant area increases is observed almost universally in studies of birds in forests (Howe 1984, Ford 1987). However, this relationship is not conclusive proof that fragmentation of forested habitat affects the composition of bird communities, as increased sample size in contiguous forest will also result in increased species richness (van Dorp & Opdam 1987). Species richness in equivalent sample areas (local species richness in the present study) is the pertinent community-level response. In Maryland, Lynch & Whigham (1984) found that local species richness was higher in small forests, i.e. that small forests have more species at a given point. It is not

unreasonable to expect this trend to be found elsewhere in the world. A large forest patch with many total species may be rich because of its diverse habitats, and therefore have relatively few species at a given point. Howe (1984) found that species richness was comparable between forest fragments and equivalent-sized areas of contiguous forest, but that species composition differed. Neither of these findings were replicated in the present study, and once again interspecific competition from noisy miners is invoked as a cause. The aggressive behaviour of noisy miners when present in colonies appears to remove so many species from a site that it is left so depauperate that even local species richness is very low. The loss of niches which results from the lower stem density, poor tree health and an open understorey does not result in a corresponding increase in available niches for open-country species, as the noisy miner expands into those niches and excludes potential invaders. It is interesting to note that the presence of an open understorey did not result in a large increase in open country species: Richard's pipit was recorded only once; the skylark (*Lauada arvensis*), striated fieldwren (*Calamanthus fuliginosus*), white-fronted chat (*Epthianura albifrons*) and banded lapwing (*Vanellus tricolor*) not at all. Having removed miner colonies from the analysis, local species richness varied negatively with moderately-sized tree density, and was higher in remnants supporting floristic group four (*E. amygdalina* dry forest - see Kirkpatrick & Gilfedder 1995). Trees 15-30 cm dbh may provide important bird habitat, although species' responses do not support this. Otherwise, neither independent variable offers easy explanation. The lack of other significant results suggests that local species richness also responds to interspecific competition, and is similar in remnants where intense competition from miners is absent. This indicates that habitat fragmentation is not a direct determinant of species richness in the Tasmanian Midlands, although it is important in the distribution of miner colonies.

The modification of analysis in the present study has altered the number of predictive variables from a wide array, with an emphasis on vegetation structure and tree health to one with a more traditional bent, towards area and isolation, which tends to be in agreement with worldwide studies (van Dorp & Opdam 1987, Askins *et al.* 1987, Bellamy *et al.* 1996). The noisy miner and its congeneric, the bell miner, appear to be without recorded parallel in the avian world, and their behaviour distorts the patterns of species richness and diversity which would be observed as a result of habitat preferences in their absence. Nevertheless, they are a real, and even a natural, phenomenon, and

cannot be summarily dismissed. Their dominance of the landscape is a result of habitat fragmentation and is an example of the way such fragmentation can alter the competitive environment without necessarily adding a new element to the overall avifauna. This process has been discussed elsewhere (Ambuel & Temple 1983) but its practical assessment has tended to be limited to parasitic bird species, such as the brown-headed cowbird in North America (Mayfield 1977, Brittingham & Temple 1983, Mark & Stutchbury 1994). The ability of miners to exclude other species is enhanced by the fragmentation process, which creates more edge-type habitat and makes it more likely that miners will be able to form colonies covering the entirety of a remnant. Management practices in most remnants inhibit the maintenance or development of an understorey species. Logging proved to be a significant factor in the presence of miner colonies. Stocking rate did not, but the measure was not especially sensitive, and exclusion of stock has been shown to aid seedling regeneration in the Midlands (Kirkpatrick *et al.* 2000).

One element which was masked by the presence of noisy miner colonies was the predictive ability of one measure of isolation: distance to the nearest forest >500 ha. This showed no significant relationships over the whole set of remnants, but in the absence of noisy miner colonies this measure of isolation showed relationships with total species richness and with % dominance by the two most abundant species. This indicates that some species do experience habitat fragmentation on the landscape scale as investigated by this research, and this is borne out by individual species' responses, as discussed below. The importance of this finding is that, even in the absence of noisy miner colonies, in remnants that are otherwise in good condition, conservation of a relatively complete avifauna relies on some level of contiguity. This would not have been apparent if analysis had not been performed on the subset of remnants excluding noisy miner colonies. The tendency for species to individual responses to isolation, which may not correspond with trends in species richness has been documented in other studies (Opdam *et al.* 1985, Lynch & Whigham 1984). These findings are important, as species sensitive to isolation may not necessarily be well served by attempts to maintain high species richness.

3.4.4 Species' responses

Not surprisingly the noisy miner also played a major role in the distribution of many species over the study area. Of thirty-seven species analysed, twenty-two were associated with low miner abundances and three were associated with high miner abundances. Once again, removal of remnants dominated by miner colonies radically altered the picture.

Most species which were intolerant of noisy miner aggression showed strong preferences for those variables which coincided with lower miner abundance: high stem density, particularly in smaller dbh classes and of smaller trees, high measures of tree health, and large remnant size. This resulted in some species displaying counter-intuitive relationships; for example, the yellow-rumped thornbill was abundant in remnants with dense understoreys, while this relationship was reversed in the absence of miner colonies. It also meant that species which could be described as ubiquitous over the whole set of remnants not only needed to have broad habitat requirements but also had to be tolerant of noisy miner aggression. Some of these anomalies were rectified by removal of noisy miner colonies from analysis. This tended to considerably reduce the number of habitat factors which differed by species presence/absence or abundance, although certain species continued to show an affinity for high stem density values. These included the spotted pardalote, yellow wattlebird and yellow-throated honeyeater, all of which might be considered indicators of remnant naturalness. The general lack of significant relationships in the absence of miner colonies indicates that most of the common species which are intolerant of miner aggression are able to occupy other remnants within the study area. This suggests that the common species perceive the habitat as relatively homogeneous, in the context of the variables measured in this study. Only five species showed positive relationships with remnant area in the absence of miner colonies. Thus, the absence of many species from small remnants appears not to be due to the lack of sufficient appropriate habitat. Rather, it relates to the competitive pressure applied by noisy miners. Isolation measures were also not important for many species, although two, the grey shrike-thrush and dusky robin, showed especially strong negative relationships with isolation. Both of these species could be considered indicative of healthy bird communities in dry sclerophyll forest. This suggests that the

commoner species perceive the habitat as relatively homogeneous, in the context of the variables measured in this study.

Dominant eucalypt species and floristic group proved to be of little predictive value, although the latter did contribute to the variance of two honeyeater species. In truth, the study was not especially well designed to detect differences in floristic preferences among bird species. These tend to be naturally patchy both temporally and spatially, and the spatial patchiness tends to be at a finer scale than that of single remnants. Mac Nally (1990) has suggested that scale of measurement is of importance in such studies. The influence of floristic elements on bird species distributions in the study area remains research worth undertaking.

The competitive pressure of noisy miner colonies has been discussed in detail, but other species associated with miner colonies, small remnants and open understorey are known to be aggressive and in some cases prey upon young birds. These include the grey butcherbird, Australian magpie and laughing kookaburra. In addition to these, Green (1983) suggested that introduction of the common starling has led to the decline of the eastern rosella in particular and Psittaciformes in general. Neither rosella species seemed particularly uncommon, neither showed any relationships with the common starling, and the eastern rosella was frequently found where the common starling was abundant. The blue-winged parrot was not common, but it is difficult to attribute any cause to this, particularly as Thomas (1979) notes its habitat preferences as being moorland, sedgeland and arable land. Nevertheless, the presence of hole-nesting introduced species (which also includes the laughing kookaburra) may be limiting access to nesting sites for all species that nest in larger tree hollows. Apart from nesting sites, the common starling uses forest remnants as roosting sites, feeding in surrounding paddocks during the day, with feeding activity increasing following ploughing. This may have a detrimental effect on native species by occupying secure sites during adverse weather conditions, but this is difficult to quantify.

Species which were at least moderately widespread could be divided into several groups in terms of their response to fragmentation and tolerance of noisy miner aggression. Firstly, there were miner-tolerant species which were unaffected by fragmentation. These included the forest raven, sulphur-crested cockatoo and common bronzewing.

Within this group a subset could be described as being particularly associated with miner colonies and preferring a fragmented landscape. These were the grey butcherbird, eastern rosella, Australian magpie, and the noisy miner itself. Secondly, there were two species which were miner-tolerant but which were adversely affected by forest fragmentation; the grey currawong and the green rosella. Thirdly, there were species that were miner-intolerant but unaffected by fragmentation in the absence of miners. These included the brown thornbill, spotted pardalote, silvereye and grey fantail. Fourthly, there were miner-intolerant species that were adversely affected by habitat fragmentation. These included the yellow wattlebird, yellow-throated honeyeater, scarlet robin and grey shrike-thrush. Finally, there were three species that were miner-intolerant but that preferred a fragmented landscape. These were the European goldfinch, musk lorikeet and common blackbird. Appendix Two contains information on the status of all species. Ignoring the categorisation of species as miner tolerant or intolerant these categories are similar to those recognised by Ford (1987) in English woods.

The classification of species above shows that the situation regarding forest fragmentation in the Tasmanian Midlands is considerably more complicated than that which has been observed elsewhere in the world (Askins *et al.* 1987, Bellamy *et al.* 1996). Only four species appear to be unequivocally advantaged by forest fragmentation, and could be considered classic edge species (abundance of species in remnant edges is further discussed in Chapter Six). A number of species would have been described as sensitive to fragmentation on initial analysis, but on closer inspection they are sensitive to competition from noisy miners. Analysis of remnants without miner colonies shows that they are distinguishable from another set of species that are sensitive to both competition and fragmentation. A few species display unusual combinations of responses, demonstrating that tolerance of miner colonies does not necessarily indicate preference for fragmented habitat and that preference for habitat fragmentation can be combined with intolerance of miner aggression.

3.4.5 Conclusion

The present above does not support the hypothesis that species richness of remnants in the study area is the result of random sampling (Connor & McCoy 1979). Species

richness was significantly related to vegetation structure, remnant landscape context and interspecific competition. The latter is considered to be the most important direct influence on species richness. Direct competition came principally from the aggressive noisy miner, but also possibly from associated edge and open-country species. The relationships between species richness, vegetation structure and remnant landscape context are considered to be a result of the habitat preferences of the noisy miner.

Only one of the common species (the common bronzewing) was randomly distributed according to the variables measured in the present study. All other common species were non-randomly distributed and were variously related to independent variables. Interspecific competition is considered to be the direct factor in the distribution and abundance of twenty-five of the thirty-seven species examined. Again, the relationships these species showed with vegetation structure and remnant landscape context are considered to be the result of the habitat preferences of the noisy miner. While interspecific competition is the major direct influence, the influence of vegetation structure and landscape context on the distribution of noisy miner colonies and therefore these factors do indirectly affect the distribution of bird species and species richness of remnants. Analysis excluding remnants excluding noisy miner colonies made it possible to discern between sensitivity to miner aggression and habitat affinities, although this did remove most of the smallest remnants from the data set.

In fact, in the absence of noisy miner colonies remnant area and isolation remain significant predictors of species richness, although large and well-connected remnants did not support more species at a given point. The maintenance of these relationships supports the hypothesis that species richness is a result of a combination of remnant landscape context and interspecific competition. The hypothesis that variability in remnant habitat *per se* determines bird species richness in the study area is rejected, although elements such as vegetation structure do play a role in affecting the competitive environment, and as in any study some factors were not measured. Four species (dusky woodswallow, fantailed cuckoo, shining-bronze cuckoo and goshawk/sparrowhawk) were randomly distributed in the absence of noisy miner colonies. One other species (brown thornbill) was only related to noisy miner abundance. Other species responded variously to remnant landscape context and habitat variability (particularly vegetation structure) and in some cases both. Thus, the

hypothesis that the distribution of species is generally determined by a combination of interspecific competition, landscape context and habitat variability is supported, although for individual species the emphasis of each of these varies, and one species could be considered to be randomly distributed throughout the study area.

The importance of habitat fragmentation in altering the balance of interspecific competition has been discussed elsewhere (Howe 1984, Ambuel & Temple 1983, Freemark & Merriam 1986). However, many studies have either not considered competition or have not emphasised its role (Lynch & Whigham 1984, Blake & Karr 1984). Elsewhere in the world the role of competition may not be as obvious as it is in the Tasmanian Midlands, where concerted aggression by one species simplifies the competitive context and has allowed assessment of competition as a factor. This competition may have masked other trends within the study area. Without analysis of remnants not dominated by noisy miner colonies the distributions of several species and trends in community-level responses would have been wrongly ascribed to vegetation structure and/or landscape context, whereas they were a result of competitive pressure.

Nevertheless, landscape context was found to be an important influence on species richness, and many species' distributions responded to landscape context and habitat variability. The importance of remnant area and isolation to species richness is a common theme worldwide (Askins *et al.* 1987, Bellamy *et al.* 1996). Other studies, which have examined species' responses, have generally found that only a proportion of these is likely to relate to area or isolation. The importance of habitat variability has been stressed for species' distributions (Lynch & Whigham 1984). Vegetation structure and other habitat variables proved to be important for species distributions in the present study, but the sensitivity of several species to remnant area and/or isolation is notable. Eleven of thirty-seven common species were considered to be sensitive to remnant landscape context. This high proportion may be due to the low overall forest and woodland cover in the Tasmanian Midlands.

The depressing effect of noisy miner colonies and the widespread distribution of such colonies has a massive impact on the avifauna of the region. Their presence in about half of all remnants 200 ha or smaller, and almost all remnants smaller than 20 ha, must be of concern. Anecdotal evidence suggests that the noisy miner is increasing in

abundance in the Midlands, and further habitat fragmentation, combined with management practices which enhance habitat for miners, is only likely to exacerbate the problem. In addition to the effect of noisy miner colonies on the bird species composition, species such as the grey shrike-thrush and dusky robin, are sensitive to habitat fragmentation even in the absence of miner colonies. These species are those which are most likely to be on the decline in the study area. These species, which are absent from remnants due to competitive behaviour from noisy miners would probably not occur in most of these remnants anyway due to their negative response to habitat fragmentation. The concluding chapter discusses the implications for remnant management in more detail.

4. Dieback in eucalypt remnants in Tasmania's Midlands

4.1 Introduction

The clearance and fragmentation of eucalypt woodlands and forests throughout Australia for agricultural purposes has not only transformed the landscape but has also been implicated in the decline of health of remaining eucalypts through a process or combination of processes known as dieback (Heatwole & Lowman 1986). Alongside tree clearance and the death of trees through old age, dieback is a major cause of tree loss in rural Australia (Wylie & Landsberg 1990).

While the causes of dieback are complicated, the symptoms are more consistent. Podger (1981) defined eucalypt dieback as the dying back of the tips or branches of the crown, which is almost invariably coincident with growth of epicormic shoots along the affected area. In some cases, the epicormic shoots may produce a new, healthy crown and the tree may recover. However, once energy reserves are exhausted no further epicormic shoots can be produced and the tree dies. In healthy eucalypt forest in a fluctuating environment some dieback and epicormic regeneration is always present, but beyond a certain level a forest may be described as dieback-affected. Dieback was not unknown to early European settlers, but the extent of the problem has increased considerably over time (Heatwole & Lowman 1986). Eucalypt death or lack of health may affect the health of other vegetation and fauna associated with them (Kile 1981). All of these problems are likely to be more pronounced in isolated remnants of eucalypt woodland and forest. Features characterising "healthy" remnants include substantial size, minimal disturbance by man or livestock, high diversity of plant and animal species and structure, and trees with good crowns, while those with high levels of dieback show the opposite trends as well as high pressure from grazing insects (Heatwole & Lowman 1986).

The causes of dieback are complicated and not entirely clear, but understanding of the processes has improved. It seems that dieback is not a single phenomenon, but is "...the

result of protracted malfunction in vital physiological processes due to the persistent action of some damaging factor or factors..." (Podger 1981). Jarrah dieback in WA is caused by a fungus disease of the roots and is reasonably well understood. Elsewhere, suggested contributory factors include defoliation (by both vertebrates and invertebrates), fungal diseases, drought, alteration of water tables, soil nutrient imbalances, soil erosion, retention of old trees without regeneration, salinity and overstocking. Probably it is a combination, possibly in synergy, and the emphasis of each one can vary spatially and temporally.

Some of the causes of eucalypt dieback in the subhumid regions of Tasmania have been examined experimentally (Kirkpatrick *et al.* 2000). The factors examined were: drought stress, defoliation by possums and stock grazing. The extent of dieback has been linked to long-term climatic changes, which saw an increase in the incidence of droughts over the last quarter of the twentieth century. However, there was no significant effect of watering trees. By contrast, at two of three sites possum-proofing mitigated dieback and stock exclusion by fencing encouraged eucalypt regeneration. Despite the lack of experimental evidence to suggest that watering may aid recovery of dieback-affected trees in the Midlands, evidence of drought as a cause of increased tree morbidity and mortality (Kirkpatrick & Marks 1985) still suggest climatic changes in subhumid Tasmania as a prospective cause of dieback. Bureau of Meteorology records for Oatlands show that only five years since 1975 have exceeded the long term mean annual precipitation, and this reduction in annual rainfall has been concomitant with a change in distribution of rainfall throughout the year, with the period of February to June receiving notably less rainfall than the long term mean.

One possible contributory factor which was not examined by Kirkpatrick *et al.* (1999) was insect damage, which has been widely cited as a cause of dieback, whether or not in concert with other factors (Old *et al.* 1981). Indeed, one of the major types of insect attack, sap-sucking, may exacerbate the effects of drought, as this process removes water from trees. Although there are difficulties in quantifying insect damage, Fox and Morrow (1983) found leaf area loss as a result of insect attack ranging from 5-44% depending on eucalypt species and location. They considered that damage from insects was greater in Australia than in the northern hemisphere, despite difficulties in making comparisons.

The orders of insect which Heatwole & Lowman (1986) listed as contributing to eucalypt dieback were Phasmatodea (stick insects), Coleoptera (beetles), Lepidoptera (moths and butterflies), Hymenoptera (ants, wasps and bees), and Hemiptera (bugs, leafhoppers, aphids, scale-insects and psyllids). Eucalypts have defences against insect attack, including volatile oils and compounds, but persistent serious defoliation can overcome these defences. Predation on insect infestations may also control or eradicate them, and birds are one of the major insectivorous groups of animals. Otvos (1979) noted that although insectivorous birds are unlikely to completely prevent insect outbreaks, bird insectivory can suppress and delay insect population build-up and thus may increase the interval between insect outbreaks, as well as accelerating the decline of an outbreak. Birds consume a high proportion of defoliating insects, commonly 40-60% but sometimes much higher (Ford 1981). "Thus, there is good evidence that birds eat most of the types of insects currently implicated in eucalypt dieback." Birds can also respond to prey increases, either by moving into an area or by preferentially taking common food. In south-eastern Australia, the psyllids, or lerp-insects, show a particularly interesting relationship with bird communities and populations.

Psyllids are sap-sucking insects which attack eucalypt foliage and which are considered to have a considerable negative impact on their host plants. These insects exude a sugary substance (honeydew) and also produce a protective coating over their nymphal stage (lerps). Both of these products are almost entirely comprised of carbohydrates. In the past it has been thought that honeyeaters have been gleaning insects from foliage and bark, when in fact they are often feeding on honeydew, lerps or manna, a sugar-rich fluid which is produced by damaged plants. All three are nectar-substitutes, and even species that consume nectar when it was available will eat manna, honeydew or lerps at other times. All bird species which consume these substance also take some insects, probably for the protein contained therein (Paton 1980). Ford (1985) considered lerps, manna and honeydew to be an unusual and characteristic component of the diet of birds in eucalypt forest. Psyllid nymphs, honeydew and lerps can comprise up to 90% of bell miner diets (Loyn *et al.* 1983). Clark (1964) considered that birds such as pardalotes may help control psyllid population levels when these remain low, but that when other factors favour psyllid outbreaks then birds were less effective.

Given that honeydew and lerps are such a rich source of food, it is not surprising that there is a considerable amount of competition for them. Wattlebirds and New Holland honeyeaters often defend rich sources of manna or honeydew, particularly from pardalotes, which are behaviourally and morphologically adapted to lerp-feeding (Paton 1980, Woinarski 1984). Bell miners and noisy miners both monopolise these food sources by their aggressive behaviour. Both miner species nest in colonies of up to several hundred individuals, which may cover up to 40 ha in the case of the noisy miner and which may completely cover isolated woodlands smaller than this size. Miner abundance is at its highest in dry sclerophyll forest with grassy ground cover. Where there is dense understorey they are not as effective at establishing colonies or excluding other birds (Dow 1977, Loyn 1987).

Establishment of miner colonies and the depressing effect that they have on bird species richness, especially of small insectivorous birds, is discussed in Chapter Three. Indiscriminate competition to defend nesting sites and food resources as practised by noisy miners and bell miners is highly unusual amongst birds (Dow 1977). Exclusion of competitors in suitable habitat allows miners to expand their feeding zones from mostly foliage to all zones. Concerted aggressive behaviour means that individuals expend relatively little energy for the benefit of a guaranteed food supply. That food supply appears to be psyllid nymphs and the lerps and honeydew that they produce. Loyn & Middleton (1981) found that bell miner distribution and psyllid infestations in the Dandenong Ranges were in almost perfect coincidence. When sawfly larvae reduced the psyllid population one study area, the miners moved elsewhere. It has been suggested that the high levels of psyllid infestation associated with miner colonies are a result of inefficient exploitation of the food source (Low 1994), but Loyn *et al.* (1983) proposed that the bell miner maintains the infestations by eating lerps but not the psyllid nymphs themselves, whereas other birds ate both. This "farming" hypothesis has been questioned (Poiani 1993, 1995) but either way the exclusion of other birds allows miners sole access to lerps and honeydew, and they do prey on them at relatively low rates. Prior to their exclusion from a psyllid infested forest, bell miners as a whole ate approximately 280 psyllids or lerps per minute, whereas following the removal of miners the invading suite of small insectivorous birds ate around 650 per minute. Within four months the infestation had been eradicated and tree health improved (Loyn 1985a). Elsewhere, removal of bell miners resulted in the eradication of psyllid infestation, and

their introduction saw psyllid levels rise once again. However, tree health did not improve in the 30 months that bell miners were absent (Clarke & Schedvin 1999). Stone (1996) found that exclusion of all birds from foliage in a bell miner colony improved leaf survival. She suggested that the miners interfered with normal psyllid regulatory factors and allowed them to reach damaging levels. However, the same study found that removal of insects by insecticide only improved tree health when in concert with removal of floristic competition by understorey thinning. These cases highlight the fact that several factors may be important in eucalypt dieback, and that treating one possible cause may not adequately address the problem.

Given the role of psyllid infestations in eucalypt dieback, and the association of miners with psyllid infestations, it would be surprising if miners showed no relationships with eucalypt dieback, and in fact they do. Ford & Bell (1981) compared the avifauna of healthy and dieback-affected forest and found radical differences in both bird diversity and total bird density. "The progression of dieback to the ultimate loss of all trees is associated with a progressive loss of birds, both of species and individuals." A healthy site averaged 48 species, and 23 birds per hectare, while a site severely affected by dieback had only 8 species and 2 birds per hectare. Small insectivorous birds were the most sensitive to dieback, despite the abundance of defoliating insects, and noisy miners were one of only four regular species at the severely affected site. Er (1997) found similar trends in eucalypt forest in the ACT and suggested indiscriminate exclusion of insectivores by noisy miners as a factor contributing to poor eucalypt health, despite conclusive proof. In Victoria, Loyn (1985a) found that small degraded forest patches were dominated by miners. Few birds fed on insects in canopies, and dieback and defoliation were widespread in these patches. Removal of noisy miners from dieback-affected forest patches resulted in invasion by the small insectivorous birds which have been considered sensitive to dieback (Grey *et al.* 1997), suggesting that it is not necessarily a lack of suitable habitat caused by eucalypt dieback that causes these birds to be absent from these patches, but competitive exclusion by miners. It seems that miner behaviour could act as a positive feedback mechanism. Miners exclude insectivores, which increases insect infestations, and therefore dieback, which reduces foliage cover and makes the habitat more open, which makes the habitat more suitable for miners and their competitive activities.

The suitability of eucalypt forest and woodland for miner colonies is also associated with vegetation structure. In Victoria, Loyn (1985b) found that noisy miners did not occur in those forest patches where intact understorey provided cover for competing birds. Likewise, Ford (1986) suggested that the loss of shrubs from the understorey has reduced the success of breeding birds, as well as other potential insect predators, such as wasps. "Removing the shrubs effectively reduces natural controls on populations of herbivorous insects." Landsberg *et al.* (1990) found that defoliation by insects, dieback and tree death were all more severe in stands of trees with pasture understoreys than those with native grasses and shrubs only lightly used by livestock. Avian diversity was also lower in degraded stands.

The bell miner is absent from Tasmania, but the noisy miner is common in the eastern third of the island in dry sclerophyll forest, eucalypt woodland and gardens (Thomas 1979, Watts 1999). Its role in reducing bird diversity and eucalypt health has not been thoroughly documented, but recent anecdotal evidence has suggested that noisy miners are on the increase, and many landowners have noticed their aggressive behaviour. This behaviour was also noted by Bosworth (1976) in dry sclerophyll remnants. These observations coincide with reports that the avifauna in agricultural areas, especially small and degraded remnants of forest, is becoming increasingly dominated by a small number of large species at the expense of small insectivorous birds. The strong association of noisy miner colonies with small remnants and with open understorey has already been established in the present study (see Chapter Three).

The dominance of bell miners and noisy miners in areas of eucalypt forest and woodland, and the fact that these colonies seems to be increasing in extent, is significant not only in terms of conservation of avifauna, but also has implications for the overall health of the fragmented natural areas within the agricultural landscape. The increased rates of dieback that are associated with miner colonies are of concern, particularly considering the possible existence of a positive feedback mechanism which maintains habitat appropriate for miners. This is especially important because birds have been shown to be effective insect predators in Australia. Ford (1985) estimated that birds ate between 55% and 70% of insect production at four sites in NSW, where miners were not present and thus can have a major influence on insect populations. Holmes *et al.* (1979) excluded birds from vegetation and showed that bird predation significantly

reduced densities of larval Lepidoptera. Clark (1964) found that small insectivorous birds might control psyllid populations when these were low, but were less effective when psyllids were very abundant. The exclusion experiments mentioned above (Loyn *et al.* 1983, Grey *et al.* 1997, 1998) all found that small insectivorous birds were able to eradicate psyllid infestations following removal of miners. Thus it would seem that a healthy community of birds may be able to contribute to improved eucalypt health.

The present study has already indicated that the presence of noisy miner colonies has a detrimental effect on bird species richness and diversity (Chapter Three). This part of the study aimed to establish any significant relationships between tree health and other independent variables (climatic, landscape, vegetation structure, management). The role of noisy miners in determining or responding to eucalypt dieback was of especial interest in the present study.

4.2 Methods

4.2.1 Dieback measures

To examine the relationships between independent variables and dieback three measures of tree health were used. The first of these was the mean dieback score according to an ordinal scale (1 = healthy tree, 2 = some dead branches in crown but mostly healthy, 3 = many dead branches in crown but reasonably healthy, 4 = many dead branches in crown, poor condition but no epicormic shoots, 5 = tree with some epicormic shoots, 6 = tree with only epicormic shoots). It was thought that this might give a subtle measure of the general health of eucalypts. Secondly, the percentage of eucalypts with epicormic shoots was used, as this obvious symptom might give a more objective measure. Finally, the proportion of dead trees (eucalypt or otherwise) was also recorded, to give an indication of overall tree health, and to investigate whether or not this followed the same patterns as measures of eucalypt health only. The trees measured were those from the transects as described in Chapter Two. Where fewer than ten trees were measured, that remnants was not included in analysis.

Independent variables as described in Chapter Two were used. Analysis was at the site scale rather than 200 m segments.

4.2.1 Data analysis

Product moment correlations were used to investigate the relationships between tree health variables and continuous independent variables, including noisy miner abundance. ANOVA and Kruskal-Wallis tests were used to investigate relationships between tree health variables and ordinal independent variables.

Subsequent analysis of floristic group against other significant variables employed ANOVA, Kruskal-Wallis tests and chi-squared analysis.

The entire data set of individual trees was amalgamated and Kruskal-Wallis tests were used to test for significant differences in mean dieback score for eucalypt species and for the dbh classes used in Chapter Two.

4.3 Results

Table 4.1 shows the differences in measures of tree health between remnants dominated by noisy miner colonies and other remnants. Dieback was significantly higher in miner colonies than in other remnants, while percentage dead stems did not differ significantly. Those variables which showed significant relationships with measures of tree health are shown in Table 4.2. Several elements stand out as especially significant predictors of tree health. Precipitation in various classes decreased significantly with eucalypt dieback and with tree death. High noisy miner abundance was associated with high levels of dieback. Due to the relationships between noisy miner abundance and precipitation discovered in Chapter Three, precipitation measures and tree health measures were tested for correlations in the absence of noisy miner colonies. The significant relationships between precipitation and dieback still hold, independent of the effect of miner colonies. Dieback and tree death decreased with increasing stem density

in the smallest dbh category; density of small trees, and dieback was highest where understorey density was low. Dieback also varied positively with several measures of canopy height. Both measures of eucalypt dieback are significantly lower for floristic group three (*E amygdalina* or *E. viminalis-E. tenuiramis* grassy woodland or dry forest on sandstone or mudstone) than for the others. Percentage of eucalypts with epicormic shoots and percentage of dead stems were both higher where fire was not used as a management tool, while percentage of dead stems was also lower where the surrounding landscape was urban. Dieback increased significantly with logging. Finally, mean dieback score decreased with remnant area, while percentage of dead stems decreased with distance to the nearest forest of 500 ha.

Testing of floristic group against other significant variables yielded three significant relationships (Table 4.3). Precipitation in the driest quarter proved to be most significant ($P < 0.01$). Group three received higher rainfall in the driest quarter than both other groups, particularly group four. Precipitation in wettest quarter also differed significantly ($P < 0.05$), with group three being intermediate between group two (which received less) and group four. Percentage logging also proved significant ($P < 0.05$). Group four had the highest levels of logging.

The dieback scores for eucalypt species and for dbh classes are shown in Table 4.4. *E. viminalis*, *E. amygdalina* and *E. globulus* suffered significantly more from dieback than did the other species. Dieback increased with increasing dbh class size.

Table 4.1. Values and standard deviations of measures of tree health for sites dominated by noisy miner colonies and other sites.

Tree health measure	Noisy miner colonies	Other Remnants	P
Mean dieback score	3.74 ± 0.69	2.94 ± 0.68	0.001
% eucalypts with epicormic shoots	39.28 ± 17.97	24.21 ± 14.07	0.01
% dead stems	20.83 ± 11.39	15.83 ± 10.87	Not significant

Table 4.2. Significant relationships between independent variables and measures of eucalypt health

Eucalypt health measure	Mean dieback score	% eucalypts with epicormic shoots	% dead stems
Variable			
Mean annual temperature[a]	*	*	*
Mean annual precipitation[a]	0.05 (-)	0.01 (-)	0.01 (-)
Mean ann. prec. (excluding miner colonies)	0.05 (-)	0.01 (-)	0.05 (-)
Precipitation in driest quarter[a]	0.001 (-)	0.001 (-)	0.001 (-)
Prec. in dri. qtr (excluding miner colonies)	0.01 (-)	0.01 (-)	0.001 (-)
Precipitation in wettest quarter[a]	*	0.01 (-)	0.05 (-)
Prec. in wet. qtr (excluding miner colonies)	0.05 (-)	0.01 (-)	0.05 (-)
Altitude[a]	*	*	*
Geology	*	*	*
Position in landscape	*	*	*
Area (ha)	0.01 (-)	*	*
Perimeter/area	*	*	*
Distance to 50 ha forest	*	*	*
Distance to 500 ha forest	*	*	0.05 (-)
Forest within 2.5 km	*	*	*
Index of local patchiness	*	*	*
Surrounding landscape	*	*	0.05
Time since isolation	*	*	*
Dominant eucalypt	*	*	*
Floristic group	0.01	0.01	*
Native species richness	*	*	*
Exotic species richness	*	*	*
Exotic species richness/native species richness	*	*	*
Significance index[a]	*	*	*
Stems/hectare	0.05 (-)	*	*
Stems <5cm dbh/ha	0.01 (-)	0.05 (-)	0.01 (-)
Stems 5-15cm dbh/ha	*	*	*
Stems 15-30cm dbh/ha	*	*	*
Stems 30-60cm dbh/ha	*	*	*
Stems >60cm dbh/ha	*	*	*
Index of vegetation structure	*	*	*
Non-eucalypts stems/ha	*	*	0.05 (-)
Eucalypt stems/non-eucalypt stems	*	*	*
Stems ≤ 6 m/ha	0.05 (-)	0.05 (-)	0.05 (-)
Maximum eucalypt canopy height	*	*	*
Mean eucalypt canopy height	0.05 (+)	*	*
Median eucalypt canopy height	0.01 (+)	*	*
Mean eucalypt height as % of max. eucalypt height	0.001 (+)	0.05 (+)	*
Basal area/hectare	*	*	*
Basal area/no. of stems	*	*	*
Overstorey recruitment	*	*	*
Understorey density	0.05	0.05	*
% trees logged[a]	0.01 (+)	0.05 (+)	*
Stocking rate	*	*	*
Fire used as management tool	*	0.05	0.01
Noisy miner abundance[a]	0.001 (+)	0.01 (+)	*

(+) or (-) indicates the direction of correlation. Absence of this symbol indicates the relationship is established by ANOVA.

[a] These variables were correlated using Spearman's rank correlation coefficient.

* = not significant.

Table 4.3. Values and standard deviations of independent variables which differed significantly according to floristic group[a]

Variable	Floristic Group 2	Floristic Group 3	Floristic Group 4	P
Precipitation in driest quarter (mm)	124.64 ± 9.99	134.0 ± 16.29	119.7 ± 5.05	0.01
Precipitation in wettest quarter (mm)	166.91 ± 11.26	179.67 ± 21.57	188.17 ± 24	0.05
Percentage logging	4.68 ± 5.04	7.71 ± 8.32	17.53 ± 14.32	0.05

[a]floristic group 2 = *Eucalyptus viminalis* grassy woodland, floristic group 3 = *E. amygdalina* or *E. viminalis-E. tenuiramis* grassy woodland or forest on sandstone/mudstone, floristic group 4 = *E. amygdalina* dry forest.

Table 4.4. Values and standard deviations of mean dieback score for eucalypt species and dbh classes using amalgamated data set

Species	Mean dieback score
E. amygdalina (n=1268)	3.18 ± 1.63
E. globulus (n=11)	3.36 ± 1.36
E. pauciflora (n=366)	2.60 ± 1.50
E pulchella (n=19)	2.26 ± 1.28
E. tenuiramis (n=438)	2.66 ± 1.71
E viminalis (n=533)	3.29 ± 1.58
	P < 0.001
dbh class	
<5 cm (n=250)	2.38 ± 1.67
5-15 cm (n=660)	2.85 ± 1.80
15-30 cm (n=774)	3.10 ± 1.59
30-60 cm (n=698)	3.19 ± 1.51
>60 cm (n=253)	3.47 ± 1.37
	P < 0.001

4.4 Discussion

The four major related sets of variables (climate, vegetation structure, floristic group and noisy miner abundance) which showed significant relationships with tree health support the argument that the causes of dieback are complex and intertwined. Although cause and effect has not been proven, several separate factors are implicated as possible causes of dieback in the Midlands, even without considering potential causes which have not been investigated in this study, such as changes to the water table and possum grazing pressure. The latter at least has already been found to be a contributing factor in the Tasmanian Midlands (Kirkpatrick *et al.* 2000). Primary cause of dieback may vary spatially and temporally.

The negative relationship of dieback with measures of precipitation supports the theory that eucalypts in the area are undergoing more sustained and intense drought pressure than has previously been the case (Kirkpatrick *et al.* 2000). Monthly rainfall means for Oatlands over the twenty years 1979-1998 were lower than the long-term means (1882-1998) for all except two months, and especially for the months of February to June (Bureau of Meteorology data). The forest and woodland habitats present in the study area were generally established prior to the shift towards a drier climate. This climatic shift, whether long or short term, may be resulting in a lower carrying capacity for trees in the Midlands, with dry sclerophyll forest being converted into woodland, and woodland being converted into grassland. Areas supporting more trees than this reduced carrying capacity could be expected to suffer higher levels of dieback and tree death. If this were the case, one would expect dieback to be more severe in areas of lower precipitation, and this is indeed observed in the Midlands. One might also expect that as some trees in a stand died more resources would be available for survivors, but experience in the field does not back this up, with some areas now full of dead trees. This may reflect the apparently synergistic effect of several contributing factors, which hasten the end of trees which are already vulnerable as a result of one factor and which make the determination of causality of eucalypt dieback so difficult.

It bears remembering from Chapter Three that precipitation is very much lower ($P < 0.001$) where noisy miner abundance is highest. Thus the problem of cause and effect is complex. Is tree health lower where rainfall is lower because miners are more abundant

in those conditions and they promote insect infestations? Or are miners more abundant where rainfall is low because the drought stress improves habitat for them? Or can either occur depending on the site? Analysis of those remnants that were not dominated by noisy miner colonies showed that the relationships between precipitation and measures of dieback and tree death persist in the absence of miner colonies. Therefore, poorer tree health where precipitation is lower is not merely a result of the affinity of noisy miners for sites with low precipitation; rather, it reflects the direct impact of climatic differences. Old *et al.* (1981) felt that insect attack was frequently an agency of dieback rather than a primary cause, which was likely to be environmental stress, and increased nitrogen in the foliage of trees under water stress has been suggested as a cause of higher insect damage on dieback-affected trees (Landsberg & Wylie 1983). Podger (1981) questioned the evidence of drought as a primary cause of dieback, and dieback has occurred in the absence of drought (Heatwole & Lowman 1986).

The presence of small trees (those in the smallest dbh class or below 6 m in height) and measures of canopy height also showed strong relationships with measures of tree health. These relationships suggest that part of the dieback problem in the Tasmanian Midlands may be due to lack of regeneration of eucalypts. Dieback score increased significantly from the smallest dbh class to the largest when the data set of all eucalypts was amalgamated. Many remnants have no regeneration. As the mature eucalypts in these remnants age they lose condition and may die. Whether this is at a higher rate than previously is not certain. However, without replacement by young trees the overall tree health of the remnant is bound to suffer. Exclusion of stock from plots in the Midlands has been shown to allow seedling regeneration (Kirkpatrick *et al.* 2000). In the long term this regeneration may replace presently adult trees and may improve overall tree health. This does not address the more direct causes of dieback and tree death, but the natural death of mature trees without replacement must be seriously considered as an element in rural tree decline in Australia. In some places the odds seem stacked against the replenishment of eucalypts, with heavy grazing pressure in climatically marginal land.

The strong relationship between eucalypt dieback and floristic group as defined by Kirkpatrick & Gilfedder (1995) is of interest, as it implies that certain communities may be more susceptible to tree decline. Examination of other significant variables against floristic groups showed that the floristic group which suffers least from dieback, group

three (*E. amygdalina* or *E. viminalis-E. tenuiramis* grassy woodland or dry forest on sandstone or mudstone) receives higher rainfall than group two in both the driest and wettest quarters, and significantly higher rainfall than group four in the driest quarter. These findings provide further evidence for the importance of drought stress in eucalypt dieback in this region.

Of the eucalypts commonly recorded in the study area, *E. viminalis* and *E. amygdalina* showed higher levels of dieback than *E. pauciflora* and *E. tenuiramis*. Lowman & Heatwole (1992) recorded variable susceptibility to dieback between eucalypt species, and it is suggested that this may be the case in the Midlands. Dominant eucalypt species of remnants showed no significant relationship with measures of tree health, indicating that the different dieback scores are not a result of geographical variation in the distribution of eucalypt species.

The negative relationships between mean dieback score and area and between percentage of trees with epicormic shoots with isolation may reflect better overall tree health in landscapes with higher overall forest cover, where external influences and disturbance may be lower. The lower proportion of dead trees where the surrounding landscape was urban probably reflects the dense stands of small trees, particularly *Acacia*, which proliferate in parts of these urban parks. These parks may also have increased nutrient levels as a result of active management or nutrient drift, and this may improve tree recruitment and survival, although Kirkpatrick & Gilfedder (1995) did not find significant differences in soil nutrients (nitrogen, phosphorus, potassium) between the edges and centres of remnants. Lack of grazing pressure and consequent soil compaction may also be favourable to trees in urban remnants. The connection between recruitment and tree health may also explain the lower proportion of eucalypts with epicormic shoots and lower proportion of dead trees where fire is used as a management tool. This is despite the fact that eucalypts are stimulated to produce epicormic shoots following fire. This measure gives no details of fire intensity or season, or the frequency of accidental fires. Nonetheless, the use of fire is known to promote regeneration of eucalypts and other sclerophyllous trees. It was also a major part of Aboriginal land management. A management regime incorporating low intensity fires similar to those used by Aborigines may also play a part in improving remnant condition.

A major aim of this study was to examine the association of noisy miners with eucalypt dieback and to speculate on the importance of these associations. The behaviour of miners and their association with psyllid infestations and dieback is well documented (Er 1997, Grey *et al.* 1997). Their impact on the avifauna and association with small remnants with an open understorey and high levels of eucalypt dieback has been demonstrated in the present study (Chapter Three). This chapter has demonstrated that dieback is more severe in remnants dominated by noisy miner colonies, and that dieback increases with miner abundance.

The theory that noisy miners maintain high levels of psyllid infestations, whether deliberately or incidentally, which in turn leads to high levels of eucalypt dieback and improves habitat for noisy miners, is an attractive one, but impossible to prove in a correlative study such as this one. Quantification of miner modification of their environment would require long term monitoring of newly-colonised areas. However, Clarke & Schedvin (1999) found that bell miners are able to establish colonies in the absence of high psyllid numbers and that following the establishment of the bell miner colony psyllid numbers rose. It is difficult to know whether noisy miners behave in the same fashion as bell miners in enhancing the growth of psyllid populations, and Clarke & Schedvin (1999) did not document improved tree health following removal of bell miners. However, the present study joins others (Er 1997, Loyn 1987) in presenting circumstantial evidence that by excluding insectivorous birds from remnants noisy miners are contributing to a decline in tree health. Levels of insect infestation were not directly measured, and causality is difficult to establish without this measurement and without long-term experimental monitoring of eucalypt health. Nevertheless, it seems that at the very least the exclusion of insectivorous birds removes one source of control of insect attacks and prevents the recovery of eucalypts from insect attack.

The present study supports the hypothesis that eucalypt dieback is associated with a combination of climatic factors, lack of regeneration and the presence of noisy miner colonies. The fact that the present study found relationships between measures of tree health and several factors which were proposed as contributing factors by Heatwole & Lowman (1986) gives further credence to their suggestion that dieback most probably has several causes which act synergistically. Insofar as dieback is caused by miner colonies, the problem in the Tasmanian Midlands may be more severe than elsewhere,

as miner colonies dominate some large remnants up to 50 ha where habitat is suitable, which is larger than estimated by Dow (1977) for Queensland. The size threshold for likelihood of miner colony presence in a habitat patch is in the region of 20-30 ha (see Chapter Three) rather than 10 ha as reported by Loyn (1987) for Victoria. Finally, the Tasmanian avifauna may provide fewer miner-tolerant species that could prey on insects. More intensive monitoring of noisy miners and of dieback in relation to noisy miner colonies may well prove fruitful in establishing direct cause of dieback in the Tasmanian Midlands. Exclusion experiments are in place on the mainland (Clarke & Schedvin 1999). Whether there is any value in replicating these in Tasmania is debatable. Suggestions for management of remnants are further discussed in the concluding chapter.

5. Seasonal movements of birds in eucalypt remnants in Tasmania's Midlands

5.1 Introduction

The ability and propensity of many species to travel considerable distances is a distinguishing feature of the class Aves. Avian movement is not restricted to dispersal of progeny from parental territories. Many species migrate. Migration is defined as regular and repeated movements of populations (Ford 1989) which therefore excludes nomadism. Nomadism is a common feature in unpredictable environments, and is quite common in Australia, notably among waterbirds (Frith 1977).

Migration may involve altitudinal, local or long-distance movements. These movements are generally from resource-poor to resource-rich areas. Nomadism may also involve evasion of bad climatic conditions and movements as a result of intra- and inter-specific competition. While the migratory habits of long-distance migrants, such as the short-tailed shearwater (*Puffinus tenuirostris*) have been apparent to humans for many years, and observers may note the appearance of species such as the eastern spinebill in urban gardens (as a response to flowering of particular plant species), the patterns of movement of many species are not clear and are often complex (Ford 1989). Keast (1968) in considering Australian honeyeaters defined seven categories of movement: latitudinal, altitudinal, none, locally moving residents, moderate nomadism, blossom nomadism, and true desert nomadism.

Bird behaviour also changes on a seasonal basis, although this can be hard to quantify. Territorial behaviour tends to be more pronounced in the breeding season, and competition may affect the distribution of species which are competing for resources. Out of the breeding season, species may be more likely to be tolerant, and mixed-species flocks occur. Due to seasonal and annual variation in weather and in food availability, foraging behaviour may also vary, as has been noted in sclerophyll forest in Tasmania (Cale 1994). It has also been observed that in Tasmania the crescent

honeyeater is almost entirely insectivorous during the breeding season, but that during winter nectar becomes an important food source (Thomas 1980).

The responses of migratory birds to habitat fragmentation can be important for conservation purposes. These species are already subject to often arduous journeys, which may significantly reduce populations. Increasing human impact on the environment can take its toll, and preservation of winter and summer habitat, as well as migratory routes is complex and difficult, as the experience of the orange-bellied parrot (*Neophema chrysogaster*) has shown (Orange-bellied parrot Recovery Team 1999). Experience in North America has shown that neotropical migrants are particularly sensitive to forest fragmentation (Robbins 1980, Ambuel & Temple 1983, Blake & Karr 1987). These species travel long distances, and therefore habitat isolation is unlikely to determine habitat selection. However, they appear to be sensitive to area of suitable habitat, quality of habitat, interspecific competition, or a combination of all three.

Very few land birds migrate to Australia from other continents, but within Australia there is a considerable degree of migration (Ford 1989). The propensity of the Australian continent to drought makes an ability to move even more valuable, and nomadism and migration are common (Slater 1995). The latitudinal range of the continent and adjacent islands also lends itself to migratory movements. Food is generally available in eucalypt forests and woodlands throughout the year but is variable in abundance both temporally and spatially, and there can be considerable seasonal and between-year variation in bird abundances as a result (Recher 1985).

Dry eucalypt forests around Australia show considerable similarity in vegetation structure and ecology and as a result they tend to support similar bird communities (Keast 1985). However, seasonal variation in bird assemblages have been documented (Lamm & Calaby 1950, Braithwaite *et al.* 1989, Loyn 1993, Arnold *et al.* 1987). Keast (1968) discovered broad correlation between amount and reliability of rainfall and the proportion of honeyeater species that moved seasonally (Australia-wide). However, where there was a steep rainfall gradient this correlation did not necessarily hold, and movements were apparently more influenced by flowering and other conditions in adjacent areas. He felt that the timing of flowering was a very important factor, and that the erratic nature of many bird movements in Australia was significantly influenced by

the irregularity of flowering. Highly nectarivorous species were much more likely to be migratory or nomadic.

European settlement of Australia may also have influenced bird migration by providing extra resources, particularly in urban areas. These can be by-products of waste disposal, such as rubbish dumps, which are frequented by silver gulls (*Larus novaehollandiae*) among others. However, another important element is the urban garden, which has spread with urban sprawl. A desire for flowering plants and the addition of nutrients can make urban gardens attractive habitat for many bird species, and the trend towards pseudo-native (Australian natives transported around the country) gardens has probably increased this attraction, although biological and ecological characteristics of birds play a part in determining whether or not they can exploit the extra resources. In south eastern Queensland, Sewell & Catterall (1998) found that urban gardens promoted a distinctive suite of native bird species, while suppressing other species which might have been present originally. The presence of extra resources may also encourage more sedentary behaviour from species that might otherwise move in search of resources.

Tasmania stands at the southward limit of movement for terrestrial birds in the Australasian realm. The island receives a number of summer visitors, although Bass Strait acts as a barrier to many species that migrate as far as southern Victoria and which might otherwise be expected to continue further south. Several summer visitors are only partial migrants; i.e. not all individuals leave the island for the winter (Ridpath & Moreau 1966). The availability of water in Tasmania is more regular than in most other parts of Australia, and local nomadism in response to drought could be expected to be a minor component of bird movements. That regularity can result in Tasmania being the recipient of wide-ranging nomads such as waterbirds when conditions are disadvantageous elsewhere (Frith 1977). The topographical diversity of the island may encourage altitudinal migration and local nomadism in response to resource availability, and this has been noted in several species, including the crescent honeyeater, eastern spinebill and yellow wattlebird (Ridpath & Moreau 1966, Keast 1968).

Elsewhere in Australia, Slater (1995) examined the responses of individual species and of guilds in Brisbane Forest Park and found that seed eaters were more common in summer, while bark-foraging invertebrate predators were more common in winter. In

south eastern Queensland, Catterall *et al.* (1998) recorded higher total bird densities and higher species richness in winter, due to the presence of winter immigrants. Differences in density and in species number between cleared land and bushland were only significant in summer, implying that the residents were most sensitive to habitat differences. In wet sclerophyll forests in Gippsland, Loyn (1993) recorded similar bird density in winter and in summer. However, species richness was higher in summer due to the presence of summer migrants. Insectivores, carnivores and "uncommon species" were all more abundant in summer, while nectarivorous honeyeaters were more abundant in winter.

Habitat fragmentation may have specific impacts on migratory bird species, and in parts of the world migrants have been especially sensitive to such fragmentation. Several studies in the North America have noted that small areas of forest support fewer bird species, mostly due to the absence of forest-dwelling, long-distance migrants (Robbins, 1980, Whitcomb *et al.* 1981, Askins *et al.* 1987). Ambuel & Temple (1983) concluded that increased competition from edge species prevents long-distance migrants from selecting small habitat fragments. Intense interspecific competition has already been observed in habitat fragments in the Midlands (Chapter Three).

Over a period of twenty-two years in the eastern states of the USA, Boulinier *et al.* (1998) found that forest fragmentation was associated with a reduction of forest bird species and also with increased temporal variability in the number of species, which was due to higher local extinction and turnover rates. This experience differs from that of some studies by Howe (1984) and Leach (1996). Over a period of twelve years, the latter found that annual species turnover in softwood scrub remnants and farmland in Queensland was higher than in extensive eucalypt forest. The former found that species assemblages in forest patches were more predictable over time than those in larger forests, although the time frame was much shorter than that of the other two studies (less than two years).

Bird community composition, therefore, would seem to be subject to several influences, including season, vegetation structure, floristics, interspecific competition and fragmentation, and these factors may well act in concert. The intention in the present study was to document changes in bird abundance and richness between years and

between seasons in the fragmented eucalypt forest and woodland habitat of the Midlands. While the arrivals and departures of summer migrants have been relatively well documented (Ratkowsky 1983), other migratory movements are less well understood and are perhaps of more interest in the context of the present study. Movements by foraging guilds were also examined to investigate patterns of movement, which may relate to resource availability. It was hypothesised that between-year variation in bird species composition would be smaller in small, open remnants, due to the role of the noisy miner in excluding many bird species. It was further hypothesised that migratory species would be more sensitive to habitat fragmentation, largely as a result of competition from resident forest and edge species. An alternative hypothesis was that migrants, which cross Bass Strait to reach Tasmania, would be less sensitive to habitat fragmentation due to their ability to cross inhospitable landscapes to reach small patches.

5.2 Methods

As described in Chapter Two, bird surveys were undertaken in Winter, Spring and Summer over a period of two years from 1996 to 1998. For the purposes of the present study, Year One comprises the samples from Winter 1996 to Autumn 1997 and Year Two comprises the samples from Winter 1997 to Autumn 1998. The large number of sites required that each set of seasonal samples be spread over a period of approximately two months. Winter samples ranged from June 30th to September 5th, Spring samples ranged from October 9th to November 29th, and Autumn samples ranged from February 17th to April 11th. To compare rainfall and temperatures of the study years with the long-term means, Bureau of Meteorology records were examined. Long term rainfall data is available from several sites within the study area. Oatlands was chosen as it has been operating since 1882. Long term temperature data are not widespread, and Launceston Airport was chosen as the closest station with lengthy continuous data (since 1939).

The non-parametric Kruskal-Wallis test was used to examine between-season and between-year differences in species richness, the Shannon-Wiener diversity index,

percentage dominance by the two most abundant species, and total bird density. The Kruskal-Wallis test was also used to examine the between-season and between-year differences in species' abundances and in abundances of foraging guilds, which are described below. Seasonal and between-year trends in species abundances were only examined for remnants in which they were recorded at least once.

To investigate seasonal movement of birds which utilise different food sources and/or foraging methods, bird species were placed into guilds and analysis of guild species richness and abundance was undertaken as per total species richness and total bird abundance. The use of guilds as a predictive tool has been questioned (Szaro 1986) but it is in widespread use (Loyn 1993, Mac Nally 1994) and at the very least allows comparisons between studies. Species were placed in foraging guilds by adapting Mac Nally's (1994) work in south-eastern Australia, an area which shares many species in common with Tasmania, and placing extra species into these guilds with reference to Thomas (1979), Watts (1999) and personal experience. Mac Nally (1994) used cluster analysis to come up with the following ten foraging guilds: sweeper; hawker; pouncer; ground carnivore; bush carnivore; bark prober; wood searcher; foliage searcher; nectarivore; and granivore. To these was added one further guild; raptors. Placement of species into guilds is listed in Appendix Two. Species were also placed into guilds according to food preferences as described by Thomas (1979). These groups were; invertebrates; vertebrates; seeds; nectar; and omnivores. Where species were placed into two of these guilds their values were used for both. Waterbirds were disregarded for the purposes of foraging and food preference guilds.

To examine whether migratory and nomadic species responded similarly to habitat fragmentation and other independent variables as described in Chapter Two, species were assigned to one of three guilds; resident, migrant or nomad, following Thomas (1979). Pearson's product-moment correlation was used to determine relationships between abundances of the migratory guilds and continuous independent variables. ANOVA or the Kruskal-Wallis test were used to determine relationships between guild abundances and ordinal independent variables. These methods are in accordance with those used to establish relationships between bird community-level responses and independent variables in Chapter Three.

Finally, in order to further examine between-year variation in avifauna, an index of similarity (SI), was calculated using the formula :

$$SI = 2c/(a+b)$$

where a = the number of species recorded in year 1, b = the number of species recorded in year 2, and c = the number of species recorded in both years (Krebs 1985 in Leach 1996). This index was tested for correlation with continuous independent variables (rank order correlation where required) and tested for significant differences according to ordinal independent variables using ANOVA or the Kruskal-Wallis test where required. This analysis was intended to determine whether independent variables played a role in determining between-year variability in the avifauna.

5.3 Results

Fifty-eight species were recorded in the first year and fifty-six in the second year, while fifty-three were common to both. Fifty species were recorded in winter, fifty-seven in spring and fifty in autumn. Forty-two species were recorded in all three seasons.

5.3.1 Climatic Data

Figure 5.1. shows the monthly mean maximum air temperature for the years 1996 to 1998 against the long term means at Launceston Airport. Figure 5.2. shows the monthly rainfall value against the long term means at Oatlands Post Office. Mean maximum air temperatures were slightly higher than the long term means in both of the summer periods of the study period, but were generally unremarkable. 1996 was the wettest year since 1985, and one of only two years post-1975 to exceed 600 mm in rainfall. The bulk of the rain was received in the summer of 1995-1996 and the autumn of 1996, prior to the commencement of sampling. 1997 and 1998 were both drier than the long term average; in the former year total rainfall was 135 mm below average.

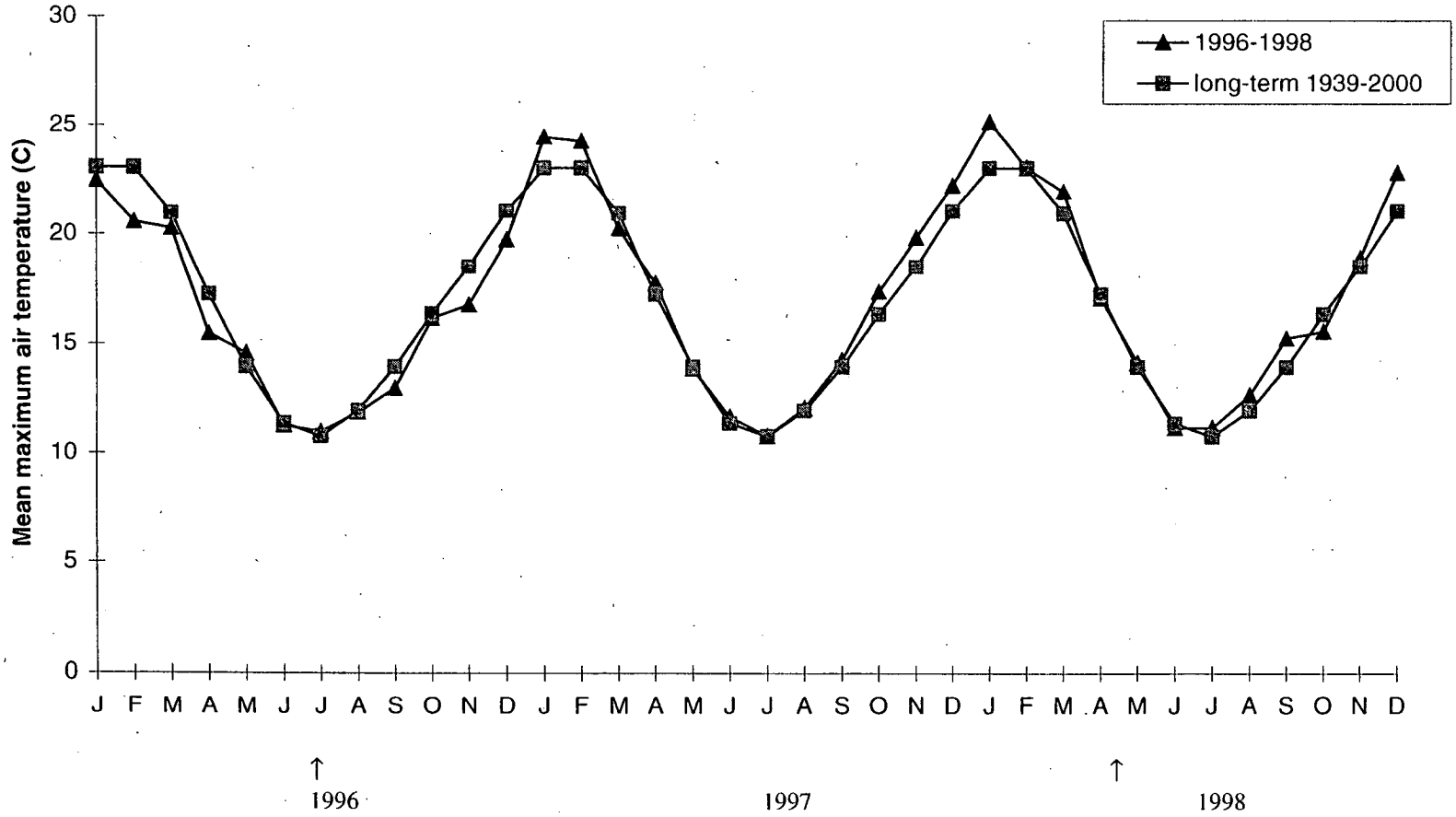


Figure 5.1. Mean maximum air temperatures for 1996-1998 and long term averages for Launceston Airport (↑...↑ indicates extent of study period).

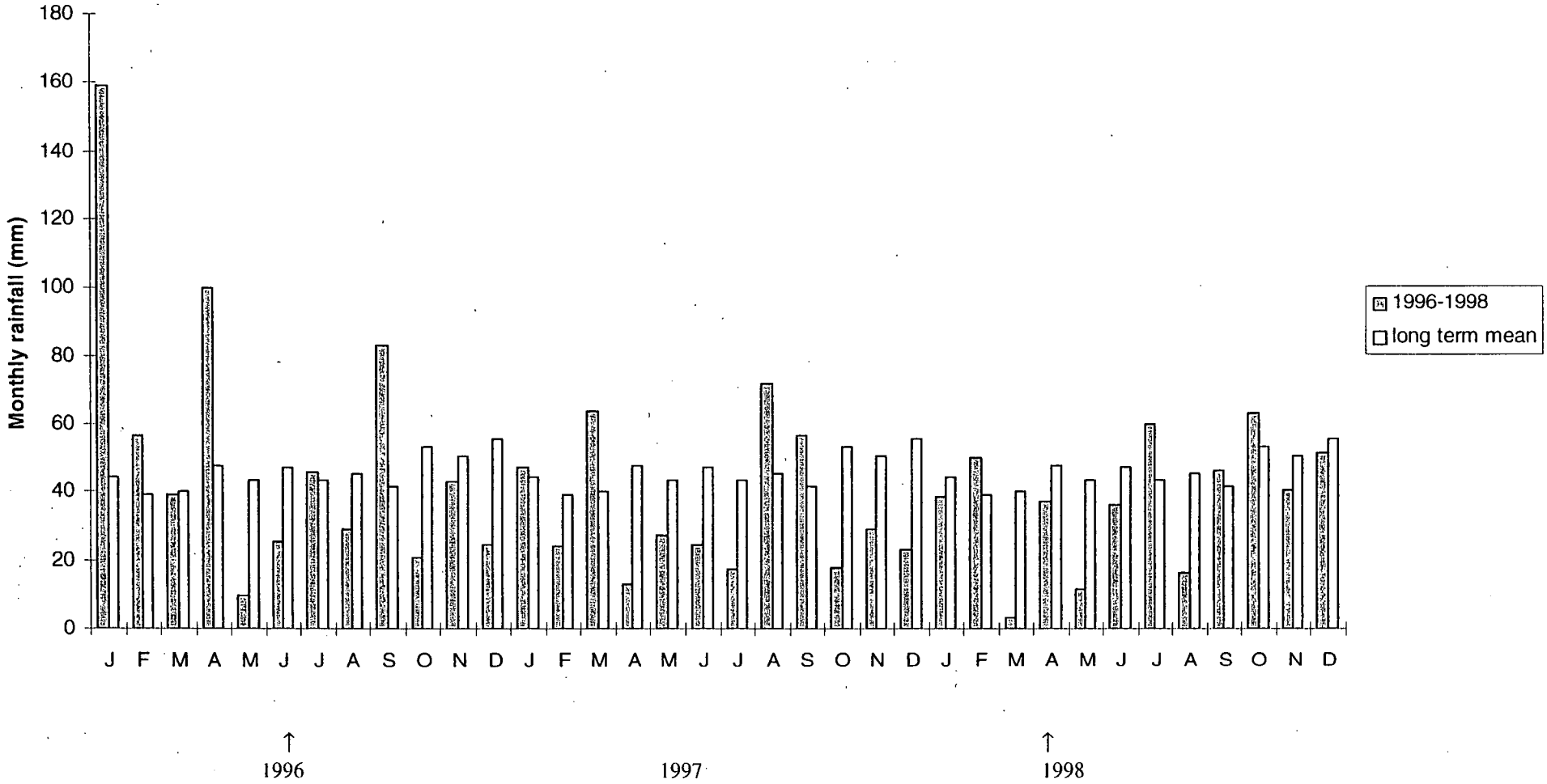


Figure 5.2. Monthly rainfall for 1996-1998 and long term averages for Oatlands Post Office. (↑...↑ indicates extent of study period).

5.3.2 Between-year variation

Table 5.1. shows the values and standard deviations for community-level responses, guild abundances and species abundances according to year as well as the significance levels of any differences. Only those species which differed significantly are presented.

Table 5.1. Between-year differences for community-level responses, and for abundance (birds/ha) of bird species and migratory, foraging and food preference guilds.

	Year 1	Year 2	P
Species richness (per survey)	12.47 ± 5.43	11.96 ± 5.08	0.508
Total bird density (birds/ha)	8.60 ± 3.92	7.34 ± 3.25	**0.002**
% Dominance of two most abundant species	51.43 ± 15.65	53.62 ± 14.96	0.162
Shannon-Wiener diversity index	0.85 ± 0.20	0.83 ± 0.19	0.431
Residents	5.70 ± 2.66	5.33 ± 2.64	0.170
Summer migrants	1.39 ± 1.34	1.17 ± 1.09	0.222
Nomads	1.93 ± 1.88	1.41 ± 1.35	**0.013**
Sweepers	0.03 ± 0.07	0.03 ± 0.06	0.988
Hawkers	0.71 ± 0.73	0.51 ± 0.53	0.189
Pouncers	0.30 ± 0.22	0.31 ± 0.22	0.655
Ground carnivores	2.80 ± 2.29	2.26 ± 2.09	**0.001**
Bush carnivores	0.26 ± 0.71	0.25 ± 0.47	**0.046**
Bark probers	0.17 ± 0.21	0.13 ± 0.16	0.433
Wood searchers	1.55 ± 1.24	1.23 ± 1.08	**0.018**
Foliage searchers	2.27 ± 2.17	1.96 ± 1.82	0.311
Nectarivores	0.10 ± 0.21	0.14 ± 0.32	0.267
Granivores	0.90 ± 0.94	1.09 ± 1.46	0.429
Raptors	0.03 ± 0.06	0.03 ± 0.06	0.418
Invertebrates	7.31 ± 3.93	5.98 ± 3.02	**0.001**
Vertebrates	0.21 ± 0.19	0.21 ± 0.20	0.686
Omnivorous	0.34 ± 0.32	0.22 ± 0.27	**0.000**
Nectar	0.30 ± 0.79	0.23 ± 0.51	0.971
Seeds	1.08 ± 1.12	1.13 ± 1.46	0.640

Table 5.2. (cont.)

	Year 1	Year 2	P
black-faced cuckoo-shrike	0.10 ± 0.17	0.04 ± 0.09	**0.001**
forest raven	0.31 ± 0.31	0.19 ± 0.24	**0.000**
musk lorikeet	0.05 ± 0.14	0.15 ± 0.32	**0.013**
striated pardalote	0.87 ± 1.04	0.49 ± 0.72	**0.000**
grey fantail	0.71 ± 0.71	0.43 ± 0.46	**0.035**
common starling	1.41 ± 1.60	1.06 ± 1.33	**0.004**

Total bird density was significantly ($P \leq 0.01$) higher in the first year. The trend towards greater abundance in the first year was also reflected by nomadic species, ground carnivores, bush carnivores, wood searchers, species which largely consume invertebrates and omnivores. Five of the six species listed were more abundant in the first year, while the musk lorikeet was more abundant in the second year. Four of the species that differed significantly were amongst the most abundant over the course of the study; the common starling, striated pardalote, grey fantail and forest raven. These were respectively the first, fourth, seventh and tenth most frequently recorded.

5.3.3 Similarity index

The similarity index was significantly correlated with only two independent variables: negatively with percentage trees removed by logging, and positively with percentage dead stems (Table 5.2). When remnants dominated by noisy miner colonies were removed the index of similarity was again correlated with two independent variables: percentage dead stems and noisy miner abundance, both of which were positive correlations (Table 5.2).

Table 5.2. Significant relationships between similarity index and independent variables

Variable	All remnants	Excluding miner colonies
% logging	0.05 (-)	*
% dead stems	0.05 (+)	0.05 (+)
noisy miner abundance	*	0.05 (+)

* = not significant, (+) or (-) indicates direction of correlation

5.3.4 Seasonal variation

Table 5.3. shows the values and standard deviations for community-level responses, guild abundances and species abundances according to season, as well as the significance levels of any differences. Species recorded only rarely are not included. Migrant status of species, as included in Appendix Two, is noted.

Table 5.3. Between-season differences for community-level responses, and for abundances (birds/ha) of bird species and of guilds.

Species[a]	Winter	Spring	Autumn	P
Species richness (per survey)	11.57 ± 4.46	12.87 ± 5.61	12.18 ± 5.58	0.340
Total bird density (birds/ha)	8.10 ± 3.70	8.38 ± 3.53	7.40 ± 3.67	0.089
% Dominance of two most abundant species	53.41 ± 14.61	53.45 ± 15.10	50.70 ± 16.21	0.191
Shannon-Wiener diversity index	0.83 ± 0.18	0.84 ± 0.19	0.85 ± 0.20	0.600
Migratory guild				
Residents	5.92 ± 2.58	5.58 ± 2.55	5.04 ± 2.77	**0.018**
Summer migrants	0.48 ± 1.13	1.56 ± 1.25	0.57 ± 0.87	**0.000**
Nomads	1.70 ± 1.87	1.21 ± 1.25	1.76 ± 1.75	**0.026**
Foraging guild				
Sweepers	0.01 ± 0.05	0.05 ± 0.08	0.02 ± 0.06	**0.000**
Hawkers	0.57 ± 0.62	0.62 ± 0.67	0.62 ± 0.66	0.937
Pouncers	0.29 ± 0.23	0.30 ± 0.22	0.31 ± 0.22	0.742
Ground carnivores	2.64 ± 2.09	2.75 ± 2.39	2.19 ± 2.11	**0.027**
Bush carnivores	0.25 ± 0.76	0.20 ± 0.33	0.32 ± 0.63	**0.038**
Bark probers	0.14 ± 0.16	0.19 ± 0.22	0.13 ± 0.15	0.563
Wood searchers	1.52 ± 1.20	1.50 ± 1.19	1.15 ± 1.10	**0.022**
Foliage searchers	1.96 ± 2.09	2.47 ± 1.97	1.91 ± 1.93	**0.019**
Nectarivores	0.09 ± 0.20	0.11 ± 0.28	0.15 ± 0.32	0.669
Granivores	0.16 ± 1.45	0.81 ± 0.75	1.03 ± 1.36	0.618
Raptors	0.02 ± 0.05	0.04 ± 0.07	0.03 ± 0.06	0.155
Food preference guild				
Invertebrates	6.60 ± 3.65	7.29 ± 3.35	6.03 ± 3.57	**0.005**
Vertebrates	0.20 ± 0.20	0.22 ± 0.16	0.22 ± 0.21	0.371
Omnivorous	0.27 ± 0.36	0.23 ± 0.24	0.33 ± 0.29	**0.006**
Nectar	0.46 ± 0.97	0.10 ± 0.27	0.25 ± 0.50	**0.003**
Seeds	1.18 ± 1.44	0.84 ± 0.77	1.31 ± 1.52	0.180

Species[a]	Winter	Spring	Autumn	P
N - yellow-rumped thornbill	0.40 ± 0.66	0.48 ± 0.63	0.42 ± 0.80	0.283
R - brown thornbill	1.30 ± 1.01	1.04 ± 0.91	1.12 ± 0.96	0.329
R - goshawk/sparrowhawk	0.001 ± 0.01	0.03 ± 0.5	0.03 ± 0.5	**0.015**
N - yellow wattlebird	0.19 ± 0.31	0.22 ± 0.34	0.20 ± 0.26	0.595
M - dusky woodswallow	0.00 ± 0.00	0.14 ± 0.19	0.28 ± 0.37	**0.000**
R - sulphur-crested cockatoo	0.43 ± 1.33	0.38 ± 0.83	0.33 ± 0.56	0.649
M - fantailed cuckoo	0.01 ± 0.04	0.06 ± 0.07	0.01 ± 0.02	**0.001**
N - European goldfinch	0.22 ± 1.05	0.19 ± 0.25	0.04 ± 0.10	**0.000**
M - shining bronze-cuckoo	0.00 ± 0.00	0.08 ± 0.09	0.003 ± 0.01	**0.000**
R - grey shrike-thrush	0.15 ± 0.15	0.18 ± 0.20	0.12 ± 0.11	0.456
M - black-faced cuckoo-shrike	0.03 ± 0.10	0.14 ± 0.18	0.04 ± 0.11	**0.000**
N - forest raven	0.25 ± 0.35	0.20 ± 0.22	0.30 ± 0.26	**0.002**
R - grey butcherbird	0.14 ± 0.16	0.12 ± 0.13	0.14 ± 0.15	0.734
M - pallid cuckoo	0.00 ± 0.00	0.06 ± 0.05	0.003 ± 0.02	**0.000**
R - laughing kookaburra	0.11 ± 0.14	0.11 ± 0.13	0.08 ± 0.12	0.131
R - brown falcon	0.03 ± 0.06	0.04 ± 0.07	0.02 ± 0.06	0.359
N - musk lorikeet	0.06 ± 0.17	0.05 ± 0.11	0.20 ± 0.37	0.054
R - Australian magpie	0.38 ± 0.37	0.32 ± 0.44	0.40 ± 0.49	0.148
M - welcome swallow	0.01 ± 0.05	0.05 ± 0.08	0.01 ± 0.03	**0.000**
R - yellow-throated honeyeater	0.52 ± 0.59	0.33 ± 0.45	0.61 ± 0.42	**0.000**
R - noisy miner	1.33 ± 1.19	1.55 ± 1.22	1.09 ± 1.13	**0.029**
N - dusky robin	0.05 ± 0.08	0.05 ± 0.09	0.09 ± 0.11	0.338
R - golden whistler	0.07 ± 0.10	0.03 ± 0.08	0.08 ± 0.11	**0.001**
N - spotted pardalote	0.35 ± 0.49	0.16 ± 0.30	0.40 ± 0.44	**0.000**
M - striated pardalote	0.37 ± 0.81	1.33 ± 1.00	0.34 ± 0.48	**0.000**
R - scarlet robin	0.16 ± 0.17	0.09 ± 0.12	0.21 ± 0.18	**0.002**
N - flame robin	0.003 ± 0.01	0.10 ± 0.12	0.02 ± 0.04	**0.000**
R - common bronzewing	0.08 ± 0.14	0.03 ± 0.07	0.07 ± 0.26	0.078
N - crescent honeyeater	0.61 ± 1.12	0.05 ± 0.20	0.24 ± 0.55	**0.000**
R - green rosella	0.22 ± 0.30	0.19 ± 0.26	0.23 ± 0.59	0.441
R - eastern rosella	0.80 ± 0.96	0.47 ± 0.46	0.75 ± 1.07	0.303
N - grey fantail	0.64 ± 0.62	0.58 ± 0.67	0.50 ± 0.55	0.404
N - grey currawong	0.04 ± 0.10	0.06 ± 0.11	0.05 ± 0.13	0.093
R - common starling	1.18 ± 1.34	1.44 ± 1.59	0.98 ± 1.48	**0.001**
R - common blackbird	0.27 ± 0.42	0.49 ± 0.82	0.15 ± 0.28	0.107
M - silvereye	0.26 ± 0.78	0.21 ± 0.33	0.33 ± 0.64	**0.037**
R - superb fairy-wren	0.68 ± 0.92	0.31 ± 0.82	0.41 ± 0.77	**0.007**

[a] Migratory status of species. R = resident, M = summer migrant, N = nomad.

All three migratory guilds showed significant relationships with season. Not surprisingly, summer migrants were most abundant in spring. Residents were most abundant in winter, and least abundant in autumn. Nomads were more abundant in both winter and autumn than in spring. Three foraging guilds which displayed significant differences in abundances were most abundant in spring (sweepers, ground carnivores and foliage searchers), while bush carnivores were least abundant in spring and wood searchers were least abundant in autumn. Invertebrate feeders were most abundant in spring, while nectar feeders and omnivores were both least abundant in spring.

Twenty of the most widespread thirty-seven species showed significant relationships with season. Many species were most abundant in spring, especially a collection of interstate migrants. However, several species described as residents by Thomas (1979) showed seasonal trends. These included the noisy miner, golden whistler, yellow-throated honeyeater and common starling. Meanwhile, some species described as nomadic did not display seasonal trends. These included the musk lorikeet, yellow wattlebird and grey currawong.

5.3.5 Migratory guilds and independent variables

Table 5.4. presents the significant relationships between abundances of migratory guilds and independent variables both for the whole collection of sites and for those sites not dominated by noisy miner colonies (i.e. not Assemblage One from Chapter Three).

As with the avifauna in general (see Chapter Three) there was considerable difference between the trends seen for all sites and those seen for sites not dominated by noisy miner colonies. Over all sites, abundance of residents and of nomads varied positively with several climatic variables. The relationships of these two guilds with precipitation in wettest quarter held for the reduced subset, as did that of nomad abundance with mean annual precipitation.

Table 5.4. Significant relationships between migratory guilds and independent variables

Variable	All sites			Excluding noisy miner colonies		
	Residents	Summer Migrants	Nomads	Residents	Summer Migrants	Nomads
Mean annual temperature	*	*	0.05 (+)	*	*	*
Mean annual precipitation	0.05 (+)	*	0.05 (+)	*	*	0.05 (+)
Precip. in driest quarter	*	*	0.05 (+)	*	*	*
Precip. In wettest quarter	0.01 (+)	*	*	0.05 (+)	*	0.05 (+)
Altitude	*	*	0.05 (-)	*	*	*
Area	0.01 (-)	0.001 (+)	0.001 (+)	*	*	*
Remnant shape	0.01 (+)	0.001 (-)	0.001 (-)	*	*	*
Distance to 50 ha forest	*	*	*	*	*	*
Distance to 500 ha forest	*	*	*	*	*	*
Area forested within 2.5 km	*	*	*	*	*	*
Index of local patchiness	*	*	*	*	*	*
Native plant species richness	0.001 (-)	*	*	*	*	*
Exotic plant species richness	*	*	*	*	*	*
Plant significance index	*	*	*	*	*	*
Stems/ha	*	0.01 (+)	0.01 (+)	*	*	*
Stems/ha (<5cm dbh)	*	0.05 (+)	*	*	*	*
Stems/ha (5-15 cm dbh)	*	0.05 (+)	0.05 (+)	*	*	*
Stems/ha (15-30 cm dbh)	*	*	*	0.05 (-)	*	*
Stems/ha (30-60 cm dbh)	*	*	0.05 (+)	*	*	*
Stems/ha (>60 cm dbh)	0.05 (+)	*	*	*	*	*
Index of vegetation structure	*	*	0.05 (-)	*	*	*
Non-eucalypt stems/ha	*	*	*	*	*	*
Eucalypt stems/non-eucalypt stems	*	*	*	*	*	*
Trees < 6m tall/ha	*	0.05 (+)	0.05 (+)	*	*	*
Maximum canopy height	*	*	*	*	*	*
Mean canopy height	0.05 (+)	*	*	*	*	*
Mean canopy height/Max. canopy height	*	*	*	*	*	*
Basal area/ha	*	*	*	*	*	*
Basal area/no. of stems	*	0.05 (-)	0.01 (-)	*	*	*
% logging	*	*	0.05 (-)	*	*	*
Mean dieback score	*	0.001 (-)	0.01 (-)	*	*	*
% epicormic shoots	*	0.01 (-)	0.01 (-)	*	*	*
% dead stems	*	*	*	*	*	*
Noisy miner abundance[a]	*	0.001 (-)	0.001 (-)	0.05 (-)	0.05 (-)	0.05 (-)
Common starling abundance[a]	*	*	0.05 (-)	*	*	*
Geology	*	*	*	*	*	*
Position in landscape	*	*	*	*	*	*
Surrounding landscape	*	*	*	*	*	*
Time since isolation	*	*	*	0.05	*	*

Table 5.4. (continued)

Variable	All sites			Excluding noisy miner colonies		
	Residents	Summer Migrants	Nomads	Residents	Summer Migrants	Nomads
Dominant eucalypt	*	*	*	*	*	*
Floristic group	0.05	*	*	0.05	*	*
Overstorey recruitment	*	*	*	*	*	*
Understorey density	*	0.001	0.001	*	*	*
Stocking rate	*	*	*	*	*	*
Fire used as management tool	*	*	*	*	*	*

(+) or (-) indicates the direction of correlation. Absence of this symbol indicates the relationship is established by ANOVA. * = not significant.

[a] The noisy miner and common starling are both classified as residents and were removed from resident abundance for analysis.

Resident abundance was negatively related to remnant area and native plant species richness, and positively related to remnant shape, stem density in the largest dbh class, mean canopy height and common starling abundance. Only the last relationship held for the reduced subset, while stem density of trees 15-30 cm dbh was negatively related for this subset. Resident abundance was also significantly higher in floristic group four (*E. amygdalina* dry forest - see Kirkpatrick & Gilfedder 1995) for all sites and for sites not dominated by noisy miner colonies.

In addition to the relationships with climate mentioned above, nomad abundance varied positively with remnant area, and some measures of stem density (total stem density , stem density < 5 cm dbh, stem density 30-60 cm dbh, and stem density of trees less than 6 m tall). Nomad abundance varied negatively with altitude, remnant shape, index of vegetation structure, basal area divided by number of stems, percentage of trees removed by logging, noisy miner abundance, common starling abundance and the two measures of dieback. It was also higher at the highest values of understorey density. Only the relationship with noisy miner abundance remained significant for the reduced subset.

Summer migrant abundance showed similar trends with nomad abundance over the whole range of sites, although significant relationships were evaluated with fewer variables. These were positive relationships with remnant area, total stem density, stem density < 5 cm dbh, stem density 5-15 cm dbh and stem density of trees less than 6 m

tall. Negative relationships were found for remnant shape, basal area divided by number of stems, noisy miner abundance and the two measures of dieback. Migrant abundance was higher at the highest levels of understorey density. For the reduced subset of sites only noisy miner abundance varied significantly (negatively) with migrant abundance.

5.4 Discussion

Caution has been suggested in interpreting seasonal differences in bird counts solely to differences in bird abundances (Loyn 1993, Boulinier *et al.* 1998). Seasonal changes may cause or coincide with changes in activity and/or vocalisation levels. For example, species may be more active in winter when searching for more scarce food resources. In spring, many species are more vocal as they seek mates. Thus, some species may be recorded more when their real abundance is the same. It is probable that some of the significant results from the present study are due to changes in the probability of detection between seasons. Nevertheless, this inherent variability must be accepted if within-year comparisons are desired. The likelihood of variation in detectability is considered to be relatively low in the habitat surveyed in the present study, in which it is considered that individuals are more likely to be detected at all times of the year than in denser forest habitats.

5.4.1 Between-year differences

The higher total bird density of the first year can be ascribed to higher productivity in that year as a result of heavier rainfall in late 1995 and in 1996. This fits in with significantly higher abundance of nomads as a group, as these species might be expected to move into an area when resources are abundant, whereas sedentariness and territoriality would be expected to inhibit the rapid increase of residents, and summer migrants would be expected to be more regular in their movements into the area, regardless of climatic fluctuations. The increase in productivity seems to have been of invertebrate populations rather than of nectar, seeds or vertebrate populations, as the invertebrate-feeding guild and omnivores were those which showed a significant increase.

Two of the species which were more abundant in the first year were summer migrants (striated pardalote and black-faced cuckoo-shrike) and their greater abundance could be partly ascribed to the fact that the winter sample of the first year stretched slightly later into the year. This may have increased observations of these species as spring arrivals may have already been present in the study area during the winter sample. Two other species which were more abundant in the first year were categorised as nomads (forest raven and grey fantail), while the classification of the common starling as a resident (Thomas 1979) does not reflect the experience in the field, where this species was observed moving between sites and following food sources in agricultural country. The abundance of all three of these species could be considered to be the result of increased productivity in the study area resulting from higher than usual rainfall. The trends in the abundance of the forest raven and common starling would have affected those of the guild of ground carnivores, which showed the most significant difference between years. The three groups which were more abundant in the first year of the present study (ground carnivores, bush carnivores and wood searchers) support the theory that invertebrates were more abundant in this year, with predators of this groups being more abundant as a result.

The similarity index showed relationships with only a very few independent variables, and only with percentage of dead stems in both sets of remnants examined. This indicates that while small remnants in poor condition may be more likely to support an impoverished avifauna (Chapter Three), this avifauna is no more predictable from year to year than the avifauna of remnants which support what would be considered a diverse avifauna. Excluding remnants dominated by noisy miner colonies did result in a positive relationship between miner abundance and the similarity index, and this could be attributed to the competitive pressure applied to non-resident species, but it is surprising that this relationship was not present over the whole data set. The positive relationship of percentage of dead stems with the similarity index suggests that species that require tree hollows as nesting sites are more likely to be faithful to remnant type, if not to individual remnants. Tree hollows may be a scarce resource in the study area. The relationship with percentage logging does not lend itself to simple explanation.

5.4.2 Seasonal differences

Total bird density was not significantly different between seasons. Fledglings were observed during spring and autumn, but were not abundant enough to affect the results. The absence of significant increases in bird abundance and species richness, despite the arrival of several migrant species is due to the concurrent absence of residents and nomads. The latter were significantly less abundant in spring. The arrival of migrants would exert extra competitive pressure in the study area, and it is postulated that nomadic species are likely to move to higher altitudes, where the populations of resident species may be lower due to a lack of resources over winter. At these higher altitudes, the productive warmer months provide sufficient resources for immigrants. These results differ from those in eucalypt forest in Victoria and on Tasmanian offshore islands, where species richness and total bird density were both higher in the breeding season than in winter and autumn (Abbott 1978).

Twenty out of thirty-seven common bird species showed significant differences in their seasonal abundances. Not surprisingly, summer migrant species showed strong seasonal trends, although only three were entirely absent in winter. Ridpath & Moreau (1966) noted that all individuals migrated in only about half of summer migrants to Tasmania, although whether overwintering of certain species (such as the fantailed cuckoo, which was recorded at Punchbowl Reserve in Launceston) reflects natural behaviour or a relatively new tendency towards overwintering as a result of increased resources from urban gardens is unclear. At least one species in which all individuals migrate, the welcome swallow, was recorded in winter. This species was recorded late in the winter sample and almost certainly represents an early arrival.

Many nomadic species showed the inverse trend to summer migrants, being most abundant in winter and autumn. Their possible summer movements are discussed below. Not all nomads showed significant seasonal differences, while four species (golden whistler, superb fairy-wren, scarlet robin and yellow-throated honeyeater) listed as residents by Thomas (1979) showed seasonal trends similar to those of the nomad guild. This may bring into question their residential status, although it may be that these species are less active or less easily observed in spring rather than actually less abundant. The superb fairy-wren, at least, is not considered to be nomadic (Rowley 1964). This

species may be less vocal and more cryptic in spring. Two other species listed as resident also showed seasonal trends. These were the noisy miner, which breeds in colonies and which may be more apparent in spring due to breeding behaviour, and the common starling, which is known for its flocking behaviour and which is thought to be another species which may not merit the status of resident.

The seasonal trends observed in migratory guilds indicate an increase in invertebrates in spring, and a resultant increase in certain foraging guilds. Sweepers, ground carnivores and foliage searchers were all more abundant in spring, while bush carnivores were less abundant in that season, and wood searchers were less abundant in autumn. These movements of foraging guilds reflect the availability of resources during the productive warmer months. Of interest is the fact that nectarivores as defined by Mac Nally (1994) showed no seasonal variation, while the guild defined by Thomas (1979) as including nectar as an important part of their diet were least abundant during winter. This reflects the inclusion of the crescent honeyeater in the latter guild, whereas it is classified as a wood searcher by Mac Nally (1994), also influencing seasonal trends of that guild. This species showed strong seasonal trends, as it feeds substantially on nectar, especially that of *Banksia marginata*, during winter but switches to invertebrates in the spring (Thomas 1980). The combination of trends showed by foraging guilds and of preference guilds suggests that the study area provides some nectar in winter, although nectarivores as a group were not especially abundant. In the warmer months invertebrates support the influx of summer migrants into the study area, while nomads tended to be most abundant in autumn, when competition may be less.

5.4.3 Migratory guilds and independent variables

Abundance of resident species showed radical differences from that of summer migrants or of nomads. Summer migrants and nomads were both less abundant in small remnants, with high levels of dieback, low stem density, an open understorey and high noisy miner abundance. Additionally, nomad abundance was positively related to higher precipitation and higher mean annual temperature. When noisy miner colonies were excluded, the only remaining relationship for summer migrant abundance was a negative one with noisy miner abundance, while the only significant relationships remaining for

nomad abundance were positive ones with two measures of precipitation and a negative one with noisy miner abundance. Increased precipitation would be related to higher productivity, and indicates that nomadic species are attracted to areas of high productivity. By contrast, summer migrants do not seem to discern between remnants in the absence of miner colonies, although the negative relationship of both guilds with noisy miner abundance signals that noisy miners exert competitive pressure on non-resident species even where they do not totally dominate a remnant.

Abundance of resident species showed contrary trends, being negatively correlated with remnant area. Resident abundance was positively associated with two measures of vegetation structure relating to tree age, and also to two measures of precipitation, which may relate to site productivity. The higher abundance of resident species in floristic group four (*E. amygdalina* dry forest) probably relate to the fact that this floristic group is located where precipitation in wettest quarter is highest, and this measure of precipitation was positively related to resident abundance. Alternatively, the relationships with precipitation measures may be incidental, and birds may rather be responding primarily to floristic group, which itself is related to precipitation. The relationships with floristic group and precipitation in wettest quarter were two of very few that remained when noisy miner colonies were excluded from analysis.

The above results clearly demonstrate that both summer migrants and nomadic species are adversely affected by habitat fragmentation, but that this is as a result of competitive pressure from resident species (notably the noisy miner) which are advantaged by the fragmentation process rather than a negative response to fragmentation itself. This is logical, given that all migrants travel considerable distances. It would be surprising to find that they had a behavioural inability to cross open ground once they had arrived. Only one summer migrant, the black-faced cuckoo-shrike, is listed as susceptible to fragmentation in Appendix Two, and abundance of this species was related only to area, not isolation, indicating that minimum territory size is probably the limiting factor. Alternatively, only two summer migrant species were not considered to be miner-intolerant. These were the welcome swallow, a species of open country which is probably too fast moving to be an easy target when it ventures into miner colonies, and the striated pardalote, which did indeed have lower abundance where miner abundance was high, but which was able to coexist with miners due to its habit of nesting in tree

hollows. Catterall *et al.* (1998) noted that winter immigrants in Queensland also tended to be tolerant of habitat fragmentation

Nomadic species are also mobile within and beyond the study area and might be expected to be similarly unaffected by fragmentation, but the behaviour of individual nomadic species may be such that they avoid long trips over open ground. Four nomadic species were considered to be susceptible to fragmentation (Appendix Two); the grey currawong, yellow wattlebird, crescent honeyeater and dusky robin. However, as a group they are unaffected except for the competitive pressure from noisy miners which occurs in the fragmented landscape. More resident species are actually considered susceptible to fragmentation (scarlet robin, yellow-throated honeyeater, grey shrike-thrush, green rosella and golden whistler). This should not be surprising, as resident species are more likely to be attached to home ranges and have less need for wide dispersal. Bentley & Catterall (1997) found similar sensitivity of residents in linear remnants. As a group they are considered to be unaffected by habitat fragmentation because of a number of resident species which increase in abundance in fragmented habitat.

Similar trends have been observed by several studies in North America which have found that neotropical migrants are most sensitive to forest fragmentation (Freemark & Merriam 1986, Blake & Karr 1984). The role of fragmentation and habitat loss have frequently been invoked as direct cause of the phenomenon, and interspecific competition has only been stressed as an important process in a few studies (Ambuel & Temple 1983).

5.4.4 Conclusion

The results do not support the hypothesis that small and isolated remnants are more predictable over time, although the present study was only conducted over two years, unlike that by Leach (1996) which was over 12 years. Neither were they less predictable, as discovered by Boulinier *et al.* (1998) in North America. Rather, a specific habitat variable (percentage of dead stems) appears to be most important in predicting the similarity of species composition between years.

Examination of relationships between migratory guilds and independent variables do support the hypothesis that summer migrants as a group are more susceptible to habitat fragmentation than residents. Nomads were also more susceptible to habitat fragmentation than residents, but for both groups the driving force is competitive pressure from species, generally residents, which are advantaged by fragmentation, rather than effective isolation of most species. These results confirm the importance of interspecific competition in determining the species composition of the study area and demonstrate the particular sensitivity of non-resident species to such competition.

6. Edge-interior differences in the avifauna of eucalypt remnants in Tasmania's Midlands

6.1 Introduction

The scale of ecological research is of great importance, and in studies of the effects of habitat fragmentation this is very much the case (Lord & Norton 1990). Many studies of birds in fragmented forest have considered the avifauna of fragments as whole, but fewer have considered variation in the avifauna within fragments. Yet habitat within fragments may not be homogeneous any more than it is in large areas of contiguous forest, and distribution of many taxa is known to vary in habitat fragments, particularly at the edges (Donovan *et al.* 1997).

Edge habitat occurs where two different habitat types meet, and is therefore common in nature, as well as being subject to the same considerations of scale as habitat fragmentation. However, research into edge effects has tended to concentrate on abrupt habitat change between anthropogenic habitats (such as logging coupes and agricultural land) and natural habitat. Habitat gradients in these edges are not necessarily steeper than more natural ones, but boundaries between habitat types may not relate to changes in the environment.; rather, they are due to human management choices, as where pasture abuts forest. Conservation issues in these areas frequently concern either the impact on or the invasion into the natural landscape by elements of the anthropogenic landscape. Edge habitat of forests in agricultural landscape may be subject to changes in microclimate and ecological process, including increased influence from the agricultural landscape, although the extent of the changes may vary (Saunders *et al.* 1991, Donovan *et al.* 1997). In terms of the avifauna this can be invasion of open-country species, or their partial use of natural habitats (i.e. roosting in forest while feeding in agricultural land). Nest predation and parasitism has been the subject of a considerable number of studies (reviewed by Paton 1994) with inconsistent results, although Luck *et al.* (1999b) found higher levels of nest predation at human-created edges in mallee eucalypt than in either naturally-occurring edges or forest interior habitat. Alternatively, there may be changes in the vegetation in habitat edges which may have an impact on the avifauna.

These may be invasion of plant species which are either attractive or repulsive to certain bird species or changes to vegetation structure which may be attractive or repulsive to bird species.

The changes mentioned above can lead to a change in the species composition of any taxonomic group in edge habitats. Sisk & Margules (1993) proposed six hypothetical species responses: habitat generalist, habitat generalist edge exploiter, habitat generalist edge avoider, habitat specialist, habitat specialist edge exploiter and habitat specialist edge avoider. In California, they found that 46-54% of bird species peaked in abundance at the edges between grassland or chaparral and oak woodland, 12-21% avoided edges, and 33-35% did not show a discernible edge response. In South Australian mallee, Luck *et al.* 1999a) classified species as open-country, edge-users and edge-avoiders.

Several species recorded in remnants in the study area forage in the surrounding landscape (generally pasture or crops) for at least part of the time, and some forage there almost exclusively, even to the extent of becoming agricultural pests. The common starling was observed feeding in large numbers in newly ploughed paddocks, and flocks of sulphur-crested cockatoos were also frequently observed feeding on seeds on the ground in paddocks. Other species utilised the open country in different ways, hirundines feeding aerially, while the Australian magpie, forest raven and others fed on invertebrates on the ground. One might expect that species making substantial use of the surrounding landscape would be more common in remnant edges, adjacent to their major foraging sites. Division of strip transects into 200 m segments, as described in Chapter Two allowed examination of the avifauna at a higher level of resolution, to determine whether the species composition was relatively uniform or spatially variable within remnants. Specifically, the intention was to establish whether bird species did show differences in abundance at the edges of the remnants as described above, and whether the distribution of noisy miner colonies was related to remnant edges.

The relationships of noisy miner colonies with small remnants in poor condition has already been well established both elsewhere (Loyn 1987, Grey *et al.* 1998) and in the present study and the intention was not to till the same ground. However, prior to the establishment of the European agricultural system the distribution of small patches of

woodland must have been restricted. It is likely that noisy miner colonies were present at the edges of extensive areas of forest and/or within those areas of forest where the vegetation structure was suitable. This is still the case today, as small areas dominated by noisy miners are present within larger forested areas, but it is not apparent whether the locations of these are determined by vegetation structure or by proximity to open country.

It was hypothesised that species identified as preferring open country would be recorded most frequently in transect segments which were adjacent to agricultural land, i.e. remnant edges. Similarly, it was hypothesised that where noisy miner colonies were present in parts of larger remnants that these would also be located at the edges of remnants. An alternative hypothesis was that the distribution of such species and of noisy miner colonies would be determined by vegetation structure.

6.2 Methods

The avifauna of the 200 m segments was classified using the same methods as used in the classification of entire remnants (see Chapter Three). The intention was not to explore in detail the nature of the bird assemblages thus established. Rather, this method was used to divide the segments into two categories; those supporting noisy miner colonies (equivalent to Assemblage One), and all others. For remnants that contained segments of both classificatory groups, ANOVA was used to establish significant differences in measures of vegetation structure and tree health. The Kruskal-Wallis test was used where data invalidated assumptions for ANOVA.

The Kruskal-Wallis test was used to establish significant differences in the abundances of commoner species between edge segments and interior segments. For this analysis, remnants were only included where a full 200 m interior segment was present, i.e. the transect was at least 600 m long. ANOVA (and the Kruskal-Wallis test where required) was used to determine differences in measures of vegetation structure and tree health between edge segments and interior segments. Bird community-level responses were similarly tested for differences between edges and interiors. These analyses were also

performed including only those remnants not entirely dominated by noisy miner colonies (i.e. excluding Assemblage One – see Chapter Three)..

6.3 Results

The 200 m segments were classified into two very distinct groups, as per classification of entire remnants (Figure 6.1). In those remnants which had been classified as supporting Assemblage One (Chapter Three) all 200 m segments were placed in a group equivalent to Assemblage One. In eighteen of twenty-nine remnants and control transects all 200 m segments were placed in the second group. This left eleven sites, which included two control transects from the same large area of forest, which had 200 m segments placed into both groups. These segments were used for further analysis to determine if there were any habitat differences between them.

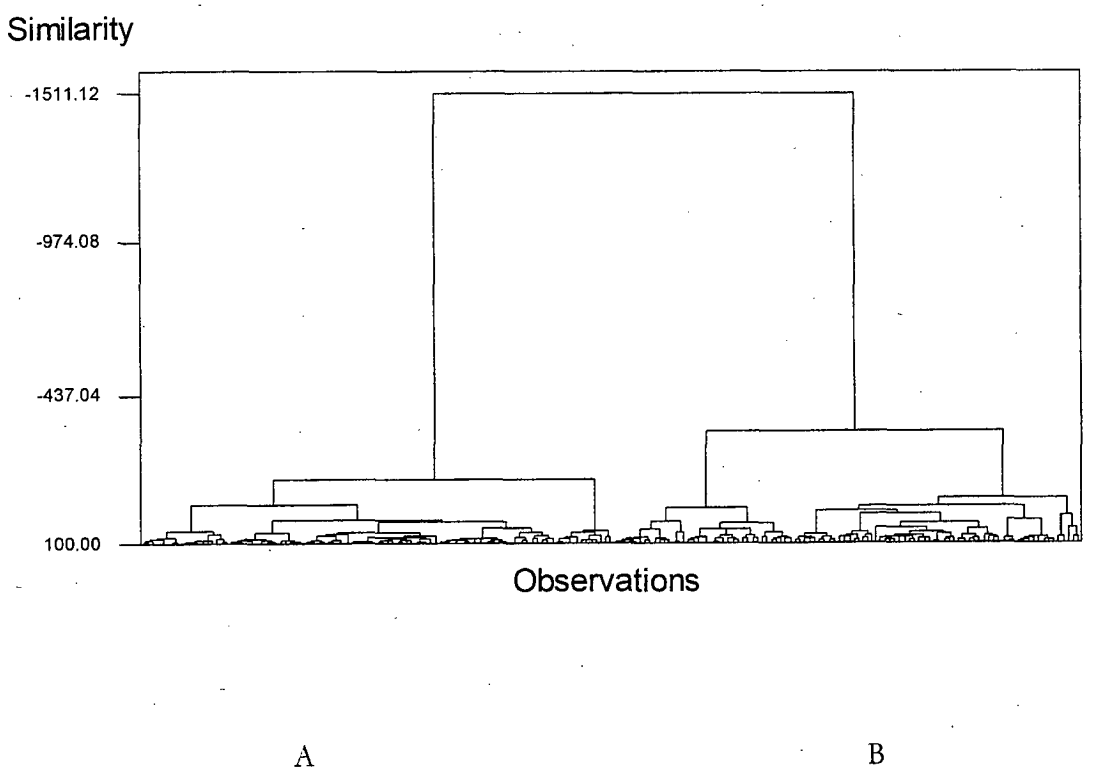


Figure 6.1. Classification of 200 m segments into avifaunal assemblages

(A = segments dominated by noisy miner colonies, B = other segments)

Table 6.1. shows the expected and observed distribution of segments supporting noisy miner colonies in edge and interior segments both for the whole range of remnants with transects 600 m or longer and for only those remnants which supported both miner colonies and other assemblages. Edge segments were significantly more likely to support noisy miner colonies ($P \leq 0.05$ for the whole data set and $P \leq 0.01$ for the reduced data set). All miner colonies which were not in edge segments were adjacent to edge segments.

Table 6.1. Contingency tables for the distribution of noisy miner colonies in edge and interior segments for (A) whole data set and (B) remnants supporting miner colonies as well as other assemblages.

Rows: Assemblage (1 = miner colony, 2 = other) Columns: Edge/Interior

(A)	Edge	Interior	All
1	32	21	53
	25.33	27.67	53.00
2	22	38	60
	28.67	31.33	60.00
All	54	59	113
	54.00	59.00	113.00

Chi-Square = 6.341, DF = 1
P-Value = 0.012

(B)	Edge	Interior	All
1	10	5	15
	6.15	8.85	15.00
2	6	18	24
	9.85	14.15	24.00
All	16	23	39
	16.00	23.00	39.00

Chi-Square = 6.624, DF = 1
P-Value = 0.010

6.3.1 Community-level responses and independent variables

Table 6.2 shows the values and standard deviations of community-level responses and independent variables for edge and interior segments. None of them differed significantly. Analysis was also performed excluding remnants entirely dominated by noisy miner colonies, and again no variables differed significantly.

Table 6.3. shows the values and standard deviations of independent variables for segments supporting noisy miner colonies and other segments within the same remnants. Community-level responses are not presented because differences in these between remnants dominated by noisy miner colonies and other remnants have already been demonstrated (Chapter Three).

Table 6.2. Values and standard deviations for independent variables and community-level responses between edge and interior segments.

Variable	Edge (n=52)	Interior (n=56)	P
stems/ha	55.98 ± 71.16	55.38 ± 45.63	0.958
stems/ha (<5cm dbh)	23.60 ± 58.71	18.09 ± 29.78	0.956
stems/ha (5-15 cm dbh)	15.46 ± 17.77	18.86 ± 18.28	0.227
stems/ha (15-30 cm dbh)	9.06 ± 6.19	9.07 ± 7.42	0.706
stems/ha (30-60 cm dbh)	5.39 ± 3.66	6.80 ± 4.51	0.103
stems/ha (>60 cm dbh)	2.21 ± 1.89	2.43 ± 1.88	0.526
index of vegetation structure	2.19 ± 3.65	1.18 ± 1.92	0.273
non-eucalypt stems/ha	31.33 ± 66.81	24.02 ± 41.40	0.606
trees <6 m tall/ha	29.33 ± 64.42	25.11 ± 37.89	0.339
maximum canopy height	18.87 ± 4.90	19.07 ± 4.76	0.909
mean canopy height	12.08 ± 4.12	12.16 ± 3.72	0.981
mean canopy height as % of maximum canopy height	62.85 ± 12.48	64.16 ± 12.58	0.502
basal area(cm^2)/ha	259160 ± 112574	285457 ± 112090	0.227
basal area ÷ no. of stems	10217 ± 8500	9284 ± 8208	0.191
mean dieback score	3.21 ± 0.84	3.23 ± 0.83	0.895
% eucalypts with epicormic shoots	32.58 ± 23.16	33.61 ± 22.53	0.739
% dead stems	21.53 ± 16.20	21.90 ± 12.74	0.611
total species richness	14.85 ± 5.19	16.46 ±4.72	0.078
dominance by two most abundance species	48.57 ± 13.56	45.62 ± 9.52	0.230
Shannon-Wiener diversity index	0.91 ± 0.17	0.96 ± 0.13	0.131
total bird density (birds/ha)	14.14 ± 5.55	14.43 ± 5.52	0.778

Table 6.3. Values and standard deviations for independent variables between noisy miner-dominated segments and other segments within the same remnants.

Variable	Miner-dominated (n=20)	Other (n=31)	P
stems/ha	42.85 ± 25.83	60.68 ± 43.02	0.131
stems/ha (<5cm dbh)	8.60 ± 20.28	22.48 ± 30.67	**0.004**
stems/ha (5-15 cm dbh)	15.05 ± 12.30	19.81 ± 16.54	0.359
stems/ha (15-30 cm dbh)	12.40 ± 7.09	9.07 ± 6.69	0.141
stems/ha (30-60 cm dbh)	4.80 ± 3.28	7.23 ± 4.72	0.104
stems/ha (>60 cm dbh)	2.00 ± 2.18	2.07 ± 1.65	0.575
index of vegetation structure	0.82 ± 0.87	0.65 ± 0.89	0.273
non-eucalypt stems/ha	12.70 ± 22.28	22.19 ± 32.85	**0.045**
trees <6 m tall/ha	13.00 ± 20.43	26.39 ± 34.25	**0.014**
maximum canopy height	17.50 ± 4.87	18.97 ± 4.23	0.265
mean canopy height	11.20 ± 2.97	11.13 ± 2.94	0.968
mean canopy height as % of maximum canopy height	63.68 ± 11.18	59.57 ± 12.54	0.263
basal area(cm^2)/ha	246523 ± 112205	269771 ± 120678	0.493
basal area ÷ no. of stems	8538 ± 7869	6787 ± 4683	0.496
mean dieback score	3.36 ± 0.79	2.99 ± 0.77	0.131
% eucalypts with epicormic shoots	37.96 ± 21.30	30.58 ± 19.48	0.318
% dead stems	24.27 ± 16.59	23.83 ± 13.00	0.841

Three measures of stem density differed significantly between miner-dominated segments and other segments within the same remnant. These were of the smallest dbh size class (<5 cm dbh), of non-eucalypt stems, and of trees less than 6 m tall. All were higher in the segments which supported assemblages other than those dominated by noisy miner colonies.

6.3.2 Species' responses

Table 6.4. shows the abundance values and standard deviations for edge and interior segments for all remnants and excluding noisy miner colonies. Only those species which differed significantly in one of the data sets are included. Over all remnants eight species were more abundant in edge segments and five were more abundant in interior habitats. For the reduced subset four significant relationships were not maintained; three species more abundant in edges and one more abundant in interior segments. The remainder of species were either too rarely recorded for effective analysis or showed no significant difference.

Table 6.4. Abundance values and standard deviations for species which differed significantly between edge and interior segments.

	All remnants			Excluding noisy miner colonies		
Species	**Edge**	**Interior**	**P**	**Edge**	**Interior**	**P**
brown thornbill	1.85 ± 1.76	2.81 ± 2.32	**0.045**	2.01 ± 1.81	3.07 ± 2.27	**0.027**
grey shrike-thrush	0.20 ± 0.24	0.32 ± 0.27	**0.034**	0.19 ± 0.24	0.32 ± 0.27	**0.024**
yellow-throated honeyeater	0.80 ± 1.10	0.95 ± 0.73	**0.045**	0.78 ± 1.10	0.95 ± 0.73	**0.030**
black-headed honeyeater	0.25 ± 0.39	1.24 ± 1.18	**0.041**	0.25 ± 0.39	1.24 ± 1.18	**0.041**
striated pardalote	1.56 ± 1.25	2.05 ± 1.20	**0.028**	1.99 ± 1.30	2.45 ± 1.04	0.078
grey butcherbird	0.23 ± 0.26	0.14 ± 0.20	**0.017**	0.17 ± 0.26	0.08 ± 0.14	0.094
welcome swallow	0.07 ± 0.09	0.02 ± 0.04	**0.041**	0.06 ± 0.09	0.04 ± 0.05	0.590
blue-winged parrot	0.17 ± 0.18	0.03 ± 0.10	**0.036**	0.15 ± 0.18	0.04 ± 0.10	0.065
Australian magpie	0.59 ± 0.62	0.28 ± 0.41	**0.003**	0.50 ± 0.64	0.08 ± 0.15	**0.000**
noisy miner	2.22 ± 1.89	1.25 ± 1.42	**0.003**	1.57 ± 1.96	0.63 ± 1.01	**0.018**
eastern rosella	1.01 ± 1.06	0.53 ± 0.74	**0.007**	0.72 ± 0.82	0.17 ± 0.37	**0.008**
common starling	2.08 ± 2.45	1.24 ± 1.30	**0.011**	1.99 ± 1.73	1.10 ± 1.10	**0.010**
common blackbird	0.33 ± 0.32	0.12 ± 0.17	**0.032**	0.41 ± 0.33	0.15 ± 0.18	**0.035**

6.4 Discussion

There were considerable differences in the avifauna of edge and interior segments which were not mirrored by changes in the vegetation variables measured, indicating that proximity to open country is a strong determinant of the distribution of several bird species and also of noisy miner colonies. Community-level responses did not show significant trends. This contradicts hypotheses which predicted higher species richness and diversity at habitat edges due to the presence of open country species as well as species typical of forests (Odum 1958, in Sisk & Margules 1993). In mallee eucalypt vegetation in South Australia, Luck *et al.* (1999a) found no increase in species diversity (Shannon-Wiener index) at forest edges in two of three sites. The third site had significantly higher species diversity at the forest edge. In the present study area interspecific competition at habitat edges is probably the cause of the absence of increased species richness at remnant edges.

The classification of 200 m segments confirmed the hypothesis that where noisy miner colonies were present in small remnants they dominated the entire remnant. In most larger remnants miner colonies, where present, occupied only a part of the remnant. These were either edge segments or contiguous with edge segments which supported miner colonies. The bird species composition of the interior of these remnants was very different.

Not only was the occurrence of miner colonies significantly higher in edge than interior segments, but where miner colonies were found in some segments of larger remnants they were associated with more open understorey. These two relationships suggest that both proximity to open country and differences in habitat are important in determining the distribution of miner colonies. The affinity of noisy miners with edge habitat is of interest, because this is not a species which forages in agricultural country to any great extent. Given that edge and interior segments showed no significant differences in the vegetation variables measured, one might have expected that the presence of miner colonies would not be related to proximity to open country. However, the remnant edges may be more degraded in ways not measured in the present study, such as grazing pressure, and this may make edges more amenable to the establishment of miner colonies. There may also be some significance in the behaviour of noisy miners, which

is described more fully in Chapter Three and also in Dow (1977). Miners establish themselves in new areas in small groups which are offshoots of existing colonies, and these are more likely to establish at the edge of a new area. Additionally, Dow (1977) noted that miners form colonies covering up to 40 ha (the present study found miner colonies dominating remnants up to 50 ha) and the integrity of colonies is very strong. Having colonised a new area, miners may simply not extend far into the interior of remnants once they have expanded to cover the area that seems to be an upper limit socially and behaviourally. Finally, there is a record of concerted conflict between a colony of noisy miners and one of bell miners in Victoria (Clarke 1984), and the presence of miner colonies at the edges of large remnants may keep one or more colonies present in a remnant separate and prevent such conflict.

The evidence presented in this chapter supports the idea that prior to European settlement miner colonies were a natural element in edge habitats without being detrimental to the regional avifauna. The presence of miner colonies in edge habitat in larger forested areas presents two conservation problems concerning the intense interspecific competition resulting from the presence of these colonies. The first, which has been well documented in this and other studies (Clarke *et al.* 1997) is the exclusion of many species from small remnants. The second is the potential that miner colonies may impede the movement of other species between larger patches to utilise interior habitat. This has not been specifically investigated, but the present study has already shown that high noisy miner abundance inhibits the use of remnants by summer migrants and nomadic species even where miner colonies do not entirely dominate a remnant (Chapter Four). Research into movement through, rather than residence in, miner colonies by other bird species would be valuable.

The abundances of the most widespread open country bird species (grey butcherbird, Australian magpie, eastern rosella and common starling) differed significantly between edge and interior segments. All four of these species, forage in open country to a considerable extent, as do two less frequently recorded species, the welcome swallow and blue-winged parrot. The other two species which were more abundant in edge segments were the noisy miner, which is discussed in detail above, and the common blackbird, the only exotic species which has successfully established itself in denser vegetation types and which was associated with dense understorey in the present study.

Its preference for edge habitat may reflect a process of continuing expansion in the study area, with initial establishment in a remnant being at habitat edges. Alternatively, it may be a result of behavioural characteristics which developed in its native Europe, where forest fragments tend to be much smaller. This species will also feed on open ground, especially in winter (H.A. Ford pers. comm.), which may explain its preference for edges. The increased abundance of open country species at eucalypt forest edges was also noted by Luck *et al.* (1999a), while the common blackbird and noisy miner may fit the definition of edge-user from that study. Habitat edges may be more productive than interior habitats, due to increased sunshine and/or nutrients or insects aggregating along edges, although this has not been certainly established in other fragmented agricultural landscapes in Australia (Majer *et al.* 2000). If this were the case, it might help to explain the increased abundance of several species at habitat edges and the propensity of noisy miner colonies to occupy remnant edges.

Only a small selection of species which prefer forested habitat showed significantly higher abundances in interior segments, although three of them, the brown thornbill, grey shrike-thrush and yellow-throated honeyeater, were typical of remnants with a healthy avifauna and the last two were adversely affected by remnant isolation (see Chapter Three). Only the black-headed honeyeater was radically less abundant (less than 50% of interior abundance) at edges and could be considered an edge-avoider *sensu* Luck *et al.* (1999a). That study also found that the spotted pardalote was an edge-avoider, which was not the case in the present study. The lack of significant results for other species that were sensitive to noisy miner aggression reflects the fact that where miner colonies were absent (and other habitat or landscape requirements were met) they were able to use all parts of the remnant.

In North America, Donovan *et al.* (1997) found that edge effects depended on landscape context, and were greater where the landscape was more fragmented. The present study was performed in a highly fragmented landscape, but the degree of local fragmentation varied between remnants. The woodlands and dry eucalypt forests of the study area comprised a reasonably homogeneous habitat, although habitat at the edge of one remnant may be similar to interior habitat in another, and different species trends may have resulted. However, the lack of significant difference in vegetation structure between interior and edge segments allows confidence that the differences measured are

due to proximity to open country *per se* rather than changes in the character of the vegetation near habitat edges. The use of 200 m segments to define edge habitats is coarser than has been suggested for edge effects (Paton 1994) but while this scale may not have allowed fine definition of differences in vegetation between remnant edges and interiors, this distance should have allowed all species using the habitat edge to have been recorded within an edge segment.

The distribution of species between edge and interior segments presented in the present study support the hypothesis that proximity to open country habitat, rather than vegetation structure, is the determining factor for the presence of most common open country species in remnant edges. Factors not measured in the present study, such as microclimate and floristics, might help explain the differences, but the most parsimonious explanation is that open country species feeding in the surrounding landscape use the nearest forested habitat for other purposes, such as roosting or nesting. Meanwhile, only a small number of species prefer the interior of remnants (only four when miner-dominated remnants are excluded. This suggests that habitat is suitable for typically forest-dwelling species regardless of its proximity to open country. This is supported by the similarity in vegetation structure between edge and interior segments. Nevertheless, when considering conservation of birds in fragmented habitats, the needs of those species which do prefer remnant interiors ought to be taken into consideration, as these habitats will be scarcer than remnant edges.

7. Grasshoppers of eucalypt remnants in Tasmania's Midlands

7.1 Introduction

As mentioned in Chapter One, the study of fauna in fragmented habitats has tended to concentrate on vertebrates, and particularly birds, while there is considerable merit in examining the responses of other taxa. There is also considerable value in making comparisons between the responses of various taxonomic groups in the same study areas, and this has been done in various places (Kitchener 1982, Robinson *et al.* 1992, Wilcox *et al.* 1986). Arthropods are the most abundant and varied phylum on the planet, present across the whole range of habitats. This variety and abundance leads to difficulty in naming and describing species, let alone understanding their ecological roles, and such understanding has lagged behind that of vertebrates and tended to concentrate on economically important species. Conservation of invertebrates has received very little attention indeed, despite their importance in many food chains and in vital ecological processes such as pollination and breakdown of organic matter, not to mention their intrinsic value (New 1992). Alteration of habitat and introduction of exotic species must inevitably have some impact on invertebrates, and insects have been described as being highly susceptible to the adverse effects of forest fragmentation (Didham *et al.* 1996). It has been suggested that invertebrates have considerable potential as "bio-indicators" (Greenslade & Greenslade 1984). Research into the biogeography of invertebrates is especially valuable considering the lack of previous studies.

The effects of habitat fragmentation on terrestrial invertebrates have been investigated in some areas, and there is an increasingly large body of work on the subject, although there has been less of an emphasis on forested habitats. Halme & Niemela (1993) found higher carabid beetle species richness in Finnish coniferous forest fragments than in contiguous forest, because smaller fragment contained more species from surrounding habitats, although some specialised forest carabid species were only found in contiguous forest. In fragmented heathland in Dorset (United Kingdom) invertebrate diversity

depended on the structure and composition of the surrounding vegetation, and invertebrate diversity at a point was negatively correlated with fragment area (Webb 1989). Panzer *et al.* (1995) considered that less than a quarter of the prairie- and savanna-inhabiting insects in the Chicago area were dependent on remnants of natural habitat. However, the rate of remnant-dependence varied between taxa, ranging from 0-2% (grouse locusts, carrion beetles, stinkbugs and treehoppers) to 83% (root-boring moths).

In Western Australia, Main (1987) concluded that trapdoor spiders persist well in small eucalypt patches, probably due to long life cycle and sedentary lifestyle, and despite low dispersal ability. In the Western Australian wheatbelt, species richness of various arthropod groups (including scorpions, spiders, isopods, cockroaches, termites, earwigs, beetles, scarabids, and ants) in *Eucalyptus salubris* remnants was more significantly related to disturbance indices than to landscape variables such as connectivity and isolation, although measures of disturbance tended to be higher in smaller remnants. (Abensperg-Traun *et al.* 1996). In a more specific study in the same area Abensperg-Traun (2000) found that even small remnants retained important functional representatives of the termite group. Rare and common species were both found in small remnants. If experiences with birds are repeated, one would expect to see higher species richness in larger remnants in good condition but with species showing separate responses to habitat variables.

In an effort to examine habitat preferences of an invertebrate group, and to make comparisons with the habitat preferences of birds, it was decided to sample grasshoppers over the same study area. Grasshoppers were chosen instead of other invertebrates because they are a visible taxon, easy to collect and relatively easy to identify. They also provide an interesting counterpoint to birds, because they tend to be perceived as creatures of open country (notably grassland and agricultural land) which utilise forest habitats secondarily. Features that support the group's general preference for life in the open include relatively large size, adaptation to saltation and biting mouthparts (CSIRO 1970). However, as has already been demonstrated for birds, many species do not experience sharp boundaries between these habitats, and may use both for different purposes, or as part of a single territory. Modification of the landscape as a result of European settlement and agricultural practices may be expected to have

reduced habitat for woodland- and forest-inhabiting species while increasing habitat for open country species.

Grasshoppers belong to the Suborder Caelifera, which is part of the Order Orthoptera (also including the Suborder Ensifera - crickets and katydids). They form a group of relatively large-bodied and generally diurnal herbivorous insects, which are familiar to humans, and of which some species are economically detrimental. This economic importance has hastened the ecological understanding of at least those species which impact on agricultural activity, such as the Australian plague locust (*Chortoicetes terminifera*). Orthopterans are common in Australia, and grasshopper genera show a high degree of endemism, probably due to a preponderance of xerophily in the suborder, which makes movement through the adjacent rainforests to the north difficult (CSIRO 1970). Although classification and phylogeny are a problem, at a species level there is a reasonable level of consistency. However, this is far from complete, and ecological studies are rare (Rentz 1996). This combination of traits means that there is considerable value in biogeographical studies of these insects. In addition, Orthopterans can be important indicators of habitat quality and as measures of human impacts (Rentz 1996). On the Hungarian steppes, Baldi & Kisbenedek (1997) concluded that grasshoppers were good indicators of habitat disturbance and naturalness.

Ecological studies of grasshoppers are rare in Australia, but in other parts the world grasshoppers have been the subject of research into habitat fragmentation and degradation, although in most cases the habitat considered has been natural grassland, fragmented as a result of agricultural activity. In fragmented Hungarian grassland habitat, Kisbenedek & Baldi (1995) found that area and distance from population sources showed relationships with grasshopper species richness. In southern Idaho, grassy sagebrush areas that had been severely disturbed by wildfires and weed invasion had higher grasshopper densities than less disturbed areas (Fielding & Brusven 1993). In the Ozark mountains of Missouri, Gerber & Templeton (1996) found that populations of the lichen grasshopper (*Trimerotropis saxatilis*), which inhabits relictual glades, are functionally isolated as a result of the nature of the intervening habitat rather than distance.

Published material relating to grasshoppers in Tasmania is sparse indeed and, as is the case elsewhere in the world, concentrates on economically important species. Tasmania differs from other parts of south-eastern Australia in being relatively free of serious outbreaks of grasshoppers (McQuillan 1982). However, the wingless grasshopper, *Phaulacridium vittatum*, can be a pest at times, to crops, gardens and pine plantations (McQuillan 1982, Forestry Tasmania 2000). Apart from this species, habitat affinities of grasshopper species are poorly-known.

The intention of this research was principally to describe the distribution of grasshopper species in remnants of eucalypt woodland and dry sclerophyll forest in the Tasmanian Midlands in order to provide some basic biogeographical information. Additionally, the aim was to examine relationships between measures of grasshopper density, species richness, and species diversity and a range of habitat variables. It was hypothesised that grasshopper species richness, and the distributions of bird species in eucalypt remnants are the result of (a) random sampling, (b) variability in remnant habitat, (c) the position of the remnant in the landscape, or (d) a combination of the last three. It was hoped that this would allow for meaningful comparisons between the responses of grasshoppers with those of birds in the same study area.

7.2 Methods

7.2.1 Grasshopper surveys

50 of the 53 sites which were sampled for birds were also sampled for grasshoppers. This was done by use of a hand-held net. Grasshoppers were stored in alcohol and later identified in the laboratory. The major sampling was done at the same time as the Autumn 1997 bird sample. Most Tasmanian grasshopper species are mature and most active at this time of year (P. McQuillan, pers. comm). An additional collection was made in Spring 1997 in an attempt to record species that mature at this time of year.

It was originally intended that a set sample time be used, but this proved to be impractical, as grasshoppers were extremely abundant at some sites, but quite sparse at

others, and no single time period would have been appropriate. A standard sweep-netting method was also not appropriate for densely vegetated sites. Instead, ten grasshoppers were collected at random at all sites as a minimum. Sampling was performed over the same transect as for birds, and it was not necessarily the first ten grasshoppers from the start of the transect which were collected. In an effort to sample the range of habitats present along the transect, sampling effort was relatively even along the transect. Due to the dominance of one species, *Phaulacridium vittatum*, at most sites, particularly where grasshoppers were abundant, more than ten grasshoppers were collected where necessary. This collection was also random and was also from the length of the transect.

Four community-level response variables were extracted from the grasshopper data. These were: total species richness; % dominance by the two most abundant species; Shannon-Wiener diversity index; and an index of density. Apart from total species richness, these variables were based on the Autumn sample. Apart from the density index the variables were determined as for the bird data. Density estimates proved to be somewhat problematical, as it was initially intended to sample for a set period of time, with the number of grasshoppers collected indicating density. As discussed above, no single time period was appropriate for all sites. Instead, grasshopper density was qualitatively assessed and placed into one of six classes: 1. Very sparse; 2. Sparse; 3. Moderately dense; 4. Dense; 5. Very dense; 6. Extremely dense. Two community-level responses which were used in analysing bird data were not used; local species richness, and mean species richness. The former was not valid as grasshoppers were collected from along the entirety of the transect and not separated into the 200 m segments. The latter was not valid because grasshoppers were only sampled once at each site (excluding the supplementary Spring sample).

Independent variables as described in the Chapter Two were used to examine the relationships between grasshoppers and habitat.

7.2.2 Data analysis

The intention was to analyse data in a similar manner to the bird data to allow meaningful comparisons to be made. To this end, Product moment correlation coefficients were used to examine relationships between continuous independent variables and the grasshopper community-level response variables. ANOVA was used to examine the relationships between variables in ordinal classes and grasshopper community-level response variables. Where necessary, variables were log- or square root-transformed to obtain a normal distribution. Tukey's method was used to detect which means differed within the ANOVA. Where transformation was not possible, rank order correlations or Kruskal-Wallis tests were performed to examine relationships between independent and response variables.

The habitat preferences of nine grasshopper species that occurred at five sites or more were examined. Presence-absence data were used for this analysis, except in the case of *Phaulacridium vittatum*, which occurred at almost all sites. This species was divided into two abundance classes, and analysis performed in the same manner as it was for presence-absence of other species. Chi-squared analysis was used to test the significance of differences between expected and observed values of variables recorded at an ordinal scale. ANOVA and Kruskal-Wallis tests were used to test the significance of differences in values of continuous variables in relation to class variables.

7.3 Results

7.3.1 Species observed

Fifteen species of grasshopper in twelve genera were collected from the sites sampled. Eight species was the maximum recorded at any one site, while four sites produced only one species. In each case this was *Phaulacridium vittatum*, which was by far the most widespread and abundant species. It was recorded from all but four of fifty sites, and in the right conditions it formed dense swarms underfoot. The next most widespread species were *Tasmaniacris tasmaniensis*, recorded from 31 sites, and *Austroicetes vulgaris*,

recorded from 30. No species apart from *P. vittatum* formed the basis of swarms, although several were moderately abundant and several were associated with *P. vittatum* swarms. *P. vittatum* comprised over 65% of all grasshoppers collected, and if there was any sampling bias it was against this species. The grasshopper fauna of the sites consisted largely of seven species, listed in order of abundance: *P. vittatum, T. tasmaniensis, A. vulgaris, Macrotona australis, Oedalius australis, A. frater* and *Goniaea australasiae.* These comprised almost 97% of the grasshoppers collected.

A full list of species recorded from the sites is contained in Appendix Three Nomenclature follows Semmens *et al.* (1992).

7.3.2 Community-level responses and independent variables

Table 6.1 displays the significant relationships between measures of grasshopper species richness, diversity and overall density and habitat variables. It is also worth noting that total grasshopper density correlated significantly with species richness ($P \leq 0.001$) in a positive direction, and % dominance by the two most abundant species ($P \leq 0.05$) in a negative direction.

Total species richness showed no significant relationships with independent variables. percentage dominance and Shannon-Wiener diversity index showed similar trends although in opposite directions. Percentage dominance was positively related to mean dieback score, percentage eucalypts with epicormic shoots and was negatively related to remnant area. Diversity showed the inverse correlations, and was also correlated positively with density of stems < 5 cm dbh. For all of these correlations, $P \leq 0.05$.

Grasshopper density showed considerably more significant relationships with independent variables. Most of these related to vegetation structure. Density correlated positively with index of vegetation structure and with basal area divided by number of stems. It correlated negatively with stems per hectare, and also with stems per hectare in the categories 5-15 cm dbh and 15-30 cm dbh. Density was also negatively correlated with mean annual temperature. ANOVA showed that density was significantly lower where the stocking rate was low than where it was either medium or high. Density was

also higher in remnants on hilltops than in those on slopes, which in turn supported higher grasshopper density than remnants on valley floors.

7.3.3 Species' distributions and independent variables

Table 6.2 displays those variables which differed significantly according to species' presence and absence. Note that only nine species were abundant enough for this analysis to be valid and that *Phaulacridium vittatum* is placed into two abundance classes.

All species showed significant relationships with one or more independent variables. *Austroicetes frater* was recorded from sites with lower mean annual temperatures and at higher altitudes, and was recorded more often than expected from remnants on sandstone supporting floristic group four (Kirkpatrick & Gilfedder 1995). This species also showed relationships with vegetation structure, being recorded where the density of non-eucalypts and of short trees was low. *Macrotona australis* also showed relationships with vegetation structure, preferring more densely vegetated sites. *Phaulacridium vittatum* showed significant relationships with measures of vegetation structure and stock rate. These were similar to those of grasshopper density, which is not surprising given that this species accounted for most of the individuals recorded. This species was also more abundant where canopy height was greater.

Russalpia albertisi was the only species which showed evidence of experiencing isolation as a result of habitat fragmentation. This species displayed a positive relationship with area and a negative one with one of the measures of isolation. Several species were linked with tree health of remnants. The most notable was *Tasmaniacris tasmaniensis*, which was negatively associated with mean dieback score and with % eucalypts with epicormic shoots. *Goniaea australasiae* was also negatively associated with % eucalypts with epicormic shoots. By contrast, *Oedalius australis* was positively associated with mean dieback score. *Urnisa rugosa* showed only one significant relationship, being recorded more frequently than expected from remnants where fire was used as a management tool.

Of the species which were recorded from only a handful of remnants, *Gastrimargus musicus* was collected in very long grass from two remnants close to the Woodstock Lagoon Wildlife Sanctuary. *Austroicetes pusilla* was recorded from three remnants in the northern Midlands as well as the suburban Boronia Hill remnant near Kingston. *Cirphula pyrrhocnemis* was recorded from two *E. pauciflora*-dominated remnants in the northern Midlands. Both of these remnants had extremely high grasshopper density and high species richness (8 and 7 species respectively). *Exarna includens* was recorded from two highly-disturbed remnants in the central Midlands as well as one in good condition near Hamilton. *Phaulacridium nanum* was recorded from the same remnant near Hamilton and one of the central Midlands remnants, as well as another highly disturbed remnant in the central Midlands. *Peakesia brunniana* was recorded from two remnants in the central Midlands.

Table 7.1. Significant relationships between independent variables and grasshopper community-level responses.

Community-level response Variable	Total species richness[a]	Species diversity	% Dominance[a]	Total grasshopper density[a]
Mean annual temperature (°C)[a]	*	*	*	0.05 (-)
Mean annual precipitation (mm)[a]	*	*	*	*
Precipitation in driest quarter (mm)[a]	*	*	*	*
Precipitation in wettest quarter (mm)[a]	*	*	*	*
Altitude (m)[a]	*	*	*	*
Geology	*	*	*	*
Position in landscape	*	*	*	0.05
Area (ha)	*	0.05 (+)	0.05 (-)	*
Shape (perimeter/area)	*	*	*	*
Distance to 50 ha forest (m)	*	*	*	*
Distance to 500 ha forest (m)	*	*	*	*
Forest within 2.5 km (ha)	*	*	*	*
Index of local patchiness	*	*	*	*
Surrounding landscape	*	*	*	*
Time since isolation	*	*	*	*
Dominant eucalypt	*	*	*	*
Floristic group	*	*	*	*
Native plant species richness	*	*	*	*
Exotic plant species richness	*	*	*	*
Exotic plant species richness/native plant species richness	*	*	*	*
Significance index[a]	*	*	*	*
Stems/hectare	*	*	*	0.05 (-)
Stems <5cm dbh/ha	*	0.05 (+)	*	*
Stems 5-15cm dbh/ha	*	*	*	0.01 (-)
Stems 15-30cm dbh/ha	*	*	*	0.01 (-)
Stems 30-60cm dbh/ha	*	*	*	*
Stems >60cm dbh/ha	*	*	*	*
Index of vegetation structure	*	*	*	0.05 (+)
Non-eucalypts stems/ha	*	*	*	*
Eucalypt stems/non-eucalypt stems	*	*	*	*
Stems ≤ 6 m/ha	*	*	*	*
Maximum eucalypt canopy height (m)	*	*	*	*
Mean eucalypt canopy height (m)	*	*	*	*
Mean eucalypt height as % of max. eucalypt height	*	*	*	*
Basal area (cm^2)/hectare	*	*	*	*
Basal area/no. of stems	*	*	*	0.01 (+)
Overstorey recruitment	*	*	*	*
Understorey density	*	*	*	*
% trees logged	*	*	*	*
Mean dieback score	*	0.05 (-)	0.05 (+)	*
% eucalypts with epicormic shoots	*	0.05 (-)	0.05 (+)	*
% dead stems	*	*	*	*
Stocking rate	*	*	*	0.01
Fire used as management tool	*	*	*	*

(+) or (-) indicates the direction of correlation. Absence of this symbol indicates the relationship is established by ANOVA or Kruskal-Wallis test.

[a] These variables were correlated using Spearman's rank correlation coefficient.

* = not significant.

Table 7.2. Grasshopper species and those independent variables which differ according to their presence or abundance

Species	Significant variables
Austroicetes frater	Altitude* (+), Index of vegetation structure* (+), Eucalypt stems/non-eucalypt stems* (+) Mean annual temperature* (-), Non-eucalypt stems/ha* (-), Trees < 6m tall/ha* (-) *Geology** (sandstone), *Floristic group** (*E. amygdalina/E viminalis-E. tenuiramis* grassy woodland)
Austroicetes vulgaris	% logging* (+) *Surrounding landscape** (pasture)
Goniaea australasiae	Mean annual precipitation* (+), Native plant species richness* (+), Exotic plant species richness* (+) % eucalypts with epicormic shoots* (-)
Macrotona australis	Stems (<5cm dbh/ha)* (+) *Understorey density** (dense)
Oedalius australis	Mean canopy height/max. canopy height*** (+), Mean dieback score* (+), *Time since isolation** (+) *Floristic group** (*E. amygdalina* dry forest)
Phaulacridium vittatum (abundance)	Index of vegetation structure*** (+),Basal area/no. of stems *** (+), Mean canopy height** (+), Mean canopy height/max. canopy height* (+), *Stocking rate** (+) Stems/ha*** (-), Stems (<5cm dbh/ha)** (-), Stems (5-15 cm dbh)/ha*** (-), Stems (15-30 cm dbh)/ha* (-)
Russalpia albertisi	Area*** (+) Perimeter/area** (-), Distance to nearest 50 ha forest* (-)
Tasmaniacris tasmaniensis	Mean dieback score** (-), % eucalypts with epicormic shoots** (-), Basal area/ha* (-)
Urnisa rugosa	*Use of fire as management tool**** (yes)

*indicates $P \leq 0.05$, ** indicates $P \leq 0.01$, *** indicates $P \leq 0.001$

(-) or (+) indicates that the variable was lower or higher where the species was present.

Underlined indicates that the relationship was determined using a Kruskal-Wallis test.

Italics indicates a chi-squared test with one or more expected value < 5.

7.4 Discussion

The absence of collection of any species from a remnant is not a definite sign that it does not occur there. Although an effort was made to collect the full range of species present during sampling, the abundance of *Phaulacridium vittatum* in places may have obscured the presence of other species. In addition, many invertebrates are more sensitive than birds to small-scale changes in habitat and thus species may not have been sampled if the transect did not intercept the appropriate microhabitat. Additionally, most sampling was done in autumn, when most species are mature and active, but some species may be more abundant in other seasons and may have been under-represented in the study. Nevertheless, several strong patterns emerged from the study and these may be assumed to be reliable and valid.

The lack of biogeographical studies of grasshoppers makes it difficult to make comparisons between the results of the present study and those elsewhere in Australia. This problem is compounded by the high degree of endemicity in Australian grasshopper genera (Rentz 1996), making comparisons with elsewhere in the world difficult. Nevertheless, the presence of only 15 species out of an Australian total of more than 700 indicates that Tasmania has a depauperate lowland grasshopper fauna when compared with mainland Australia. It may also reflect the fact that the study sites were not the optimum habitat for the Suborder Caelifera in general. Grasshoppers as a group are considered creatures of open country and the Midland fauna may well be richer than is evident in eucalypt-dominated remnants.

In terms of absolute numbers the grasshopper fauna was dominated by species whose preferred habitat is open country and pasture. *Phaulacridium vittatum* was by far the most abundant species, and this has been noted as a species of pasture (McQuillan 1982). Other species recorded which Rentz (1996) described as preferring open country were *Oedalius australis, Austroicetes frater, A. vulgaris,* and *Gastrimargus musicus.* The dominance of a single species, *Phaulacridium vittatum,* has parallels in North America. Quinn & Walgenbach (1990) found that one species, *Opeia obscura,* comprised 41% of all individuals recorded from South Dakota grassland. *P. vittatum* and *A. vulgaris,* in particular, formed dense swarms in open grassy remnants where the division between surrounding pasture and remnant was not great at ground level. The latter is known to

utilise bare ground at the base of pasture trees (P. McQuillan pers. comm.). Some species more typical of forest habitat were widespread and even abundant, notably *Tasmaniacris tasmaniensis*, but these species did not form the dense swarms which were observed where remnant ground cover approximated pasture.

7.4.1 Community-level responses

Grasshopper species richness showed no significant relationships with any of the independent variables measured in the present study, but was very strongly positively correlated with grasshopper density, which is a common experience in ecology. This may imply that eucalypt remnants are not ideal habitat for several of the species recorded, but that when conditions in the remnant approximated their preferred habitat (i.e. relatively open county) they tended to be present. One would expect that this would involve similarity to grassland or similar open habitat, but no relationship was discovered between species richness and measures of stem density, which might have been expected to give an indication of this. Species richness may be responding to habitat gradients which were not measured in the present study. Otte (1976) found that grasshopper species richness in North and South American deserts was correlated with plant species richness, and Quinn & Walgenbach (1990) found evidence for relationships between grasshopper species and grass species richness in South Dakota prairies. In South Dakotan mixed-grass rangeland, meanwhile, Quinn *et al.* (1991) found that grasshopper species richness was positively associated with soil characteristics, notably sand content.

Unlike species richness, both derived measures of grasshopper species diversity did show significant relationships with independent variables. Both the Shannon-Wiener diversity index and dominance by the two most abundant species were significantly correlated with remnant area and with two measures of tree health. These could be considered to represent remnant condition, as large remnants tend to be in a more natural state and dieback is a sign of deterioration, as discussed in Chapter Four. When combined with the association between grasshopper density and low stem density and open understorey as well as high stock rate, these results are consistent with findings elsewhere in the world which have found that in disturbed habitats grasshopper

abundance increases while measures of diversity decrease. Grazing and other forms of disturbance have been shown to result in higher grasshopper density and lower diversity (Fielding & Brusven 1993, Baldi & Kisbenedek 1997) and affect grasshopper community structure (Quinn & Walgenbach 1990). Grazing by domestic stock in native woodlands of Western Australia has been shown to reduce native plant species richness and increase the proportion of exotic plant species and may result in the loss of functional plant groups and affect ecological functioning (Milchunas & Lauenroth 1993, Pettit *et al.* 1995). Fire frequency has also been implicated as an influence on grasshopper species richness and composition (Gibson & Hulbert 1987, Evans 1988). The use of fire as a management tool was significant only for the distribution of one species in the present study and was not related with any community-level measures.

7.4.2 Species distributions

The distribution of only one of the commoner species, *Russalpia albertisi*, was related to remnant area and isolation and could be described as adversely affected by habitat fragmentation. This species is widespread in many open habitats, but is most abundant in subalpine open shrubland and herbfield (P. McQuillan pers. comm.). The distributions of other species, with the exception of *Urnisa rugosa*, were positively or negatively associated with one or more variables that could be considered indicators of remnant condition. Four species, *Austroicetes frater, A. vulgaris, Oedalius australis* and *Phaulacridium vittatum* preferred remnants which were relatively similar to pasture, while three species, *Goniaea australasiae, Macrotona australis* and *Tasmaniacris tasmaniensis* preferred more "forest-like" remnants. These last three species were especially likely to be recorded if there were elements of heath in the ground cover. *G. australasiae* is a dead eucalypt leaf mimic, and its association with forests is to be expected. These trends are in keeping with habitat preferences described by Rentz (1996) and McQuillan (pers. comm.).

7.4.3 Comparisons with avifaunal trends

Not surprisingly, trends in species richness and diversity for birds and for grasshoppers were slightly different, which could be attributed to different biological and ecological characteristics. Grasshopper species richness was not significantly related to any independent variables measured, while bird species richness was related to several, although exclusion of noisy miner colonies from analysis reduced those relationships markedly. By contrast, grasshopper density was sensitive to independent variables, while bird density did not differ significantly in the absence of noisy miner colonies. This suggests that, unlike birds, grasshoppers have access to extra resources in more open and disturbed remnants. These may be food sources, such as pasture plant species that have invaded the remnants. Another possibility is the shelter provided by sparse trees. The two groups did show some similarities in their responses to habitat fragmentation. Both groups showed differences in species composition in remnants that could be described as in poor condition, in which the open country elements of the fauna became dominant. Individual species in both groups showed strong relationships with measures of habitat structure, but only one grasshopper species was affected by remnant area and isolation, whereas several bird species were sensitive to habitat fragmentation. This may reflect the influence of scale. Grasshoppers may be affected by patch size, but at a smaller scale than birds.

The similarities in response to habitat fragmentation can be attributed to similarities in biology and ecology. Dispersal ability has been described as vital to the survival of arthropod species in fragmented landscapes (den Boer 1990) and grasshoppers, in general, are a vagile group. However, many Tasmanian species are flightless and these taxa will be less vagile than winged genera such as *Austroicetes*. *Phaulacridium vittatum,* is unusual in being dimorphic for wing length. Other arthropod groups with poorer powers of dispersal might be expected to react more negatively to habitat fragmentation. Grasshoppers as a group are creatures of open country, and several species utilise both the forested and pastoral elements of the landscape. For both birds and grasshoppers, habitat fragmentation and associated land management has benefited the elements of the fauna that prefer open country. The response of grasshoppers as a group appears to be to habitat variables whereas birds are affected by the change in competitive balance that arises from the presence of noisy miner colonies. The impact

of such competition (if any) is not known for grasshoppers. By whatever means, several bird and grasshopper species typical of open country were the most abundant and were prominent in degraded remnants, while others were sensitive to changes in habitat and were restricted to remnants in good condition.

7.4.4 Conclusion

A closer inspection of microhabitat and measurement of groundcover characteristics might have better informed trends of species richness and of species' distributions. For example, habitat preferences in two species of grasshopper in Colorado were determined by the thermoregulatory behaviour of the species (Anderson *et al.* 1979). However, in the context of the present study grasshopper species richness is not related to any of the variables measured and the hypothesis that it is random throughout the study area is supported. By contrast, individual species were non-randomly distributed, and the results of this study support the hypothesis that variation in habitat determines the distribution of most species. Only one species experienced isolation as a result of the fragmentation of eucalypt forest in the study area. This is not to say that the grasshopper avifauna is unaffected by habitat fragmentation. Kruess & Tscharntke (1994) felt that habitat fragmentation affected natural enemies more than their phytophagous hosts. As a result, populations of invertebrates may be affected by the responses of their predators and parasites to habitat fragmentation.

The relationships of diversity measures and grasshopper density with habitat variables which indicate remnant condition are mirrored by species composition of remnants in various condition. These findings mirror those of other studies, where degraded habitats supported higher grasshopper density and lower diversity (Baldi & Kisbenedek 1997, Fielding & Brusven 1993). The former counselled against placing too much stress on species richness as an indicator of quality, but the present study suggests that remnant condition has an important role to play in species composition in the Tasmanian Midlands, and that remnants in poor condition are likely to be dominated by a single species, *Phaulacridium vittatum.*

Grazing history is known to affect vegetation structure of forests and woodlands (Milchunas & Lauenroth 1993, Pettit *et al.* 1995) and has also been shown to affect total

grasshopper density in the Canadian prairie sites (Capinera & Sechrist 1982). Grass coverage has been found to affect the abundance of obligate grass feeders, obligate forb feeders and mixed grass and forb feeders (Quinn *et al.* 1991). Many of the remnants in the study area are grazed, although the extent of grazing pressure varies. Management of remnants for grazing is likely to have a major impact on the vegetation structure through suppression of tree recruitment and promotion of grasses and pasture-associated species such as clover. This will in turn benefit grasshopper species which are associated with pasture. If it is to consider the conservation of grazing-sensitive grasshopper species, management of remnants must consider the exclusion of grazing and the establishment of non-grassy ground cover.

The present study has uncovered some strong trends and described habitat preferences of a group of invertebrates which has been largely ignored in Tasmania. It is hoped that this information will be built upon in future, both in terms of further research into the biogeographical characteristics of grasshoppers and examination of more invertebrate groups in habitat fragments in the Midlands.

8. Conclusion

8.1 Introduction

Bird assemblage and species distributions, and trends in bird species richness in the Tasmanian Midlands present a strong theme of an altered competitive environment radically altering the avifauna. The role of interspecific competition in influencing avifauna of fragmented habitats has been largely overlooked elsewhere in the world (but see Ambuel & Temple 1983). In Australia, the noisy miner has been well established as a major player (e.g. Loyn 1987, Grey *et al.* 1998), and in Tasmanian dry sclerophyll forest and woodland remnants it is paramount. The noisy miner bestrides this study like a colossus, and provides a cohesive story regarding the bird species composition of the study site. Habitat fragmentation, rather than simple habitat loss (Andren 1994) has had a major impact on the avifauna of the study area. While grasshopper species' distribution followed different distribution gradients, there was nevertheless a strong theme of division between remnants in good condition and those which were degraded and disturbed. The present chapter considers the patterns of bird and grasshopper species richness in terms of the theories outlined in Chapter One, and discusses the management implications of the results of the present study.

8.2 Theories of species richness

Patterns of bird species richness in the study area do not conform to the major theories regarding the relationships between species and area; namely, random sampling, island biogeography and habitat diversity. A significant increase in species richness with increasing area and decreasing isolation was observed in the present study, although species richness per unit area was not significantly related to any independent variables. The species compositions of small remnants are not random samples of the species found in larger forests. They form a discrete assemblage that varies little between remnants. The make-up of this assemblage is not a result of immigration and extinction as predicted by the theory of island biogeography. Several summer migrants return to the area every summer, and nomadic species also move within and beyond the study area. Selection of forest fragments by migratory species has been shown to be

complicated (Villard *et al.* 1995), and these species cannot be considered to be recolonising isolated remnants every year in the same way that true islands may be colonised. Although habitat heterogeneity has been found to be an important predictor of species richness in other studies (Kitchener 1982, Boecklen 1986), trends in species richness in the study area do not seem to be strongly related to habitat variability. Stem density was a significant predictor of species richness, but the relationships of species richness with vegetation structure are related to interspecific competition. Both area *per se* and remnant habitat variability are strongly associated with species richness, but these associations are a result of their influence on the distribution of noisy miner colonies.

In the absence of noisy miner colonies the trends of species richness did show trends which support the theory of island biogeography. Species richness was higher in large remnants and lower in isolated remnants, suggesting that in the absence of the strong influence of miner competition, metapopulation dynamics may be determining species richness in the study area. Certainly, several resident and nomadic species were sensitive to either remnant isolation or area or both. In keeping with other findings (Lynch & Whigham 1984, Opdam *et al.* 1985), the responses of individual species showed a wide range of responses to habitat fragmentation. Interspecific competition was again the dominant factor, but several species appear to be affected by isolation as a result of the fragmentation of woodland and forest in the Tasmanian Midlands.

The major source of interspecific competition in the Tasmanian Midlands is the presence of noisy miner colonies, although other species may play an important role, as has been documented in other studies (Ambuel & Temple 1983). The nature of miner colonies has been described both in the present study and elsewhere (Dow 1977, Grey *et al.* 1997). These colonies may entirely dominate remnants up to 50 ha where the vegetation structure is appropriate. In larger remnants miner colonies may form in areas of open woodland around the remnant edge. In Queensland, Catterall *et al.* (1997) noted that the species compositions of small remnants tended to be similar to that of forest edges as well as that of forest with open understorey. Increases in noisy miner abundance were associated with all three factors. Similar trends were observed in the Tasmanian Midlands, supporting their suggestion that habitat selection and interspecific competition were important elements in species' distributions rather than extinction and recolonisation. Remnants not totally dominated by miner colonies supported a much

more varied avifauna, and species tended to respond individually to subtle habitat gradients.

The role of the noisy miner emphasises the importance of local factors in ecological considerations and the need for empirical studies as well as universal theory. Review of the literature suggests that the noisy miner, and its congeneric, the bell miner (*Manorina melanophrys*), is an extraordinary factor, and therefore habitat fragmentation in an area where miner colonies occur would be likely to have a different outcome than in an area where they do not, even if the general habitat types were similar. These local factors must be factored into predictions of impacts of habitat fragmentation. Despite the strong local nature of the results of the present study, the experiences in the Tasmanian Midlands may help to inform those elsewhere. The influence of an altered balance of interspecific competition on the avifauna of this fragmented landscape is likely to be replicated in other parts of the world, albeit probably not to the extent documented in the present study. In fact, the expansion of the brown-headed cowbird (*Molethrus ater*) into north-eastern North America as a result of forest fragmentation does have some parallels with the Tasmanian experience. While the process involved is brood parasitism which may be reducing songbird breeding below replacement levels (Brittingham & Temple 1983) rather than direct interference, the concept of certain species being advantaged by habitat fragmentation at the expense of others is similar. Biotic interactions may have been overlooked in other studies in favour of area- and isolation-dependent changes in habitat that may not be valid. The present study has benefited from the obvious impact of miner colonies, which has drawn attention to the importance of interspecific competition in fragmented habitats.

While the grasshopper fauna showed some different trends from the avifauna, and grasshopper species richness was random in the study area, this taxon displayed a similarly sharp division between remnants which were populated by large numbers of *Phaulacridium vittatum* and associated pasture species, and those which supported a range of forest and woodland species. Vegetation structure predicted the division of remnants and the distribution of several species, although this pertained to how similar remnants were to pasture. Distribution of grasshopper species was in almost all cases related to habitat variation rather than remnant landscape context. Remnants which are heavily grazed and with open grassy understorey are effectively extensions of the surrounding

agricultural landscape and probably do not differ in species composition from nearby paddocks, whereas those with denser understorey and non-grassy ground cover do support a different grasshopper fauna from that of the surrounding paddocks.

8.3 Comparisons between bird and grasshoppers

The different responses of birds and grasshoppers to habitat fragmentation can be ascribed to biological and ecological differences between the groups. These include differences in food preferences, life cycle and scale of perception of landscape. Interspecific competition was not examined in grasshoppers. Nevertheless, some similarities between the taxa are notable. In each taxon, several species typical of open country were the most abundant and were prominent in degraded remnants. Only a limited number of species were experiencing a fragmented landscape, which may reflect the vagility of both groups, but several species were sensitive to changes in habitat and were restricted to remnants in good condition. Some species in both groups showed unique responses in the range of habitat variables to which they were related.

8.4 Management Implications

Temperate eucalypt woodlands have suffered disproportionately from the introduction of European agricultural methods to Australia and woodland remnants face conservation problems separate from those affecting birds or grasshoppers (Yates & Hobbs 1997). Nevertheless, management plans which are intended to improve conditions of eucalypt remnants in the Tasmanian Midlands must consider fauna as well as flora. The results from the present study indicate that management policies which are designed to maintain and restore natural bird and grasshopper faunas are unlikely to conflict with those designed to improve the condition of vegetation.

The inhibiting effect of a dense understorey, especially of non-eucalypts, on noisy miner abundance and behaviour suggests that even smaller remnants may sustain a healthy suite of bird species. Maintenance of large remnants of good quality is imperative, but astute management of smaller remnants will improve their value for a range of birds.

Small size as a negative influence on the avifauna may be mitigated by the presence of a dense understorey. The condition of small remnants for birds may be improved by allowing regeneration or even by active planting of understorey species. Many Australian bird species have evolved in a naturally patchy landscape and are thus resilient to human-induced fragmentation (Ford & Barrett 1995). Effective improvement of the agricultural landscape for such species may mostly involve improving the condition of eucalypt remnants, although other species will require extensive areas of forest and woodland to persist.

Experimental evidence elsewhere in Australia suggests that physical removal of individual noisy miners is an effective method of removing the species from eucalypt remnants (Grey *et al.* 1997) and that this removal leads to a reinvasion of bird species which are absent as a result of intense competition from miner colonies. Evidently there is a critical mass which is required for a miner colony to dominate an area, and therefore not every individual must be removed to disrupt the functioning of a colony and to halt the competitive impact of miners (Grey *et al.* 1997). The fact that noisy miners tend to colonise new areas in small groups means that areas cleared of noisy miners are unlikely to be immediately reinvaded by this species. The removal process has inherent problems. Capturing individuals is a time- and labour-intensive process, and relocation of captured individuals may involve introduction of the species into a new area. The social organisation of the species means that the introduced individuals are unlikely to be accepted into areas already occupied by miners (Clarke & Schedvin 1997). Killing captured individuals is effectively culling a species that has adapted well to the agricultural landscape, and this presents ethical questions. Finally, concentration on removal of miners from degraded areas may detract from efforts to rehabilitate these areas in other ways. If this does not occur then removal of miners is only a short term solution, as eventually they will recolonise such areas.

Removal of noisy miners might be considered a viable management process if it were considered a direct threat to bird species that are of importance for conservation. On the mainland the noisy miner is considered a threat to the regent honeyeater (*Xanthomyza phrygia*) (Low 1994). In the subhumid parts of Tasmania the swift parrot and the forty-spotted pardalote (*Pardalotus quadragintus*) are at risk. The former did not show any relationship with the distribution of miner colonies and the latter was not

recorded in the study area but removal might be appropriate if miners were felt to be affecting these species elsewhere in Tasmania. Removal might also be considered, as a means to improve eucalypt health, although the cost involved would seem to be prohibitive on a large scale. However, several factors have been implicated in eucalypt dieback, and the presence of noisy miner colonies has not been conclusively proven to be a direct cause. Long-term studies on the effect of noisy miner removal on eucalypt health are in place on the mainland, with equivocal results thus far (Clarke & Schedvin 1999). A similar experiment in the Tasmanian Midlands may prove to be of value in future. At any rate, removal is unlikely to be successful unless in concert with other forms of management, as dieback is considered to be a result of multiple causes. If long term maintenance or recovery of an area is the objective, improvement of the condition of remnant vegetation is likely to precede improvement in its avifauna, rather than vice versa.

A further problem with concentrating on the problem of noisy miners in the fragmented rural landscape is that it deflects attention from other problems that are equally worthy of attention. Fragmentation of forest and woodland habitat in the Tasmanian Midlands is detrimental to the avifauna, and although the presence of noisy miner colonies may exacerbate this problem, it remains a problem even in their absence. The effects of grazing on vegetation has been linked to the abundance of many passerine species in the Western Australian wheatbelt (Arnold & Weeldenburg 1998) and inhibition of regeneration by sheep grazing is considered to be detrimental to the maintenance of a healthy avifauna in Victorian buloke remnants (Watson *et al.* 2000).

Several species in the present study are adversely affected by remnant isolation and habitat loss. Small remnants which were not dominated by noisy miner colonies were able to support a wide range of small insectivorous birds. This has also been shown to be the case on the mainland (Grey *et al.* 1998). However, the present study has shown that a number of species requires either large forested areas or a high degree of habitat connectivity or both. Proactive conservation measures for these species must include preservation of larger remnants. These species are those which are most likely to be on the decline in the study area and they should be monitored for population decline. Management processes intended to benefit such species are likely to be at the expense of native species, such as the grey butcherbird or noisy miner, which have benefited

from the development of an agricultural landscape. It seems unlikely that such processes will put at risk any of these species given their affinity with the agricultural landscape.

Not all small forest remnants are inexorably destined to be avifaunally depauperate, and studies elsewhere have emphasised the importance of small remnants in avifauna conservation (Barrett *et al.* 1994). The preservation of remnants which have a relatively dense understorey component and the conversion of other remnants to such a vegetation structure may result in a general improvement in the grasshopper and bird fauna which require forested habitat. However, the vegetation type which is thought to have considered to have covered large areas of the Midlands prior to European settlement is open grassy woodland (Fensham 1989), and birds and grasshoppers would have been adapted to large areas of this vegetation type. Small remnants of such vegetation are likely to be dominated by noisy miner colonies, while the relationship of miner colonies with the edges of large remnants indicates that conservation of large contiguous areas would be more likely to maintain a natural bird species composition as well as approximating pre-European vegetation types. Grasshoppers which are uncommon in degraded remnants are more likely to require dry sclerophyll forest rather than woodland, and conversion of grassy woodland to pasture may not have had a severe impact on most open country species, although at least one species, *Brachyexarna longipennis*, has possibly become endangered as a result of pastoralisation (P. McQuillan pers. comm.).

Decline of eucalypts in Tasmania's agricultural regions is an issue of considerable concern, and several possible causes have been invoked, some of which imply that noisy miners either enable or accelerate insect attack on trees by excluding other insectivorous birds which may control infestations (Stone 1996, Loyn 1995). At the very least noisy miner colonies are associated with eucalypt dieback and they may well inhibit recovery, while at the worst there may be a positive feedback mechanism which dooms eucalypts in noisy miner colonies to continued decline. The death of foliage improves habitat for noisy miners and makes removal of other birds more efficient. This potential positive feedback mechanism is not the only factor contributing to eucalypt decline. The present study has discovered a strong association between rainfall and eucalypt dieback. This may be linked with a prolonged dry period since the late 1970s. Additionally, dieback increases with eucalypt age, and lack of eucalypt regeneration may be exacerbating tree

decline in the Midlands. Very probably all three factors contribute to eucalypt dieback, in combination with other factors not measured in the present study.

Conservation of endangered plant species has been shown to be unrelated to general remnant condition (Kirkpatrick & Gilfedder 1995) and may require specific management which must take precedence over the maintenance or recreation of what is considered to be a natural state. Elsewhere, management plans for improvement of eucalypt health in remnants will most likely aid the conservation of bird and grasshopper species which are adversely affected by the habitat fragmentation which has a occurred as result of agricultural practices. Whether or not they are a direct cause, presence of miner colonies is a strong indicator of poor tree health, and efforts to conserve and/or restore remnants in good condition in terms of avifaunal composition will probably also lead to conservation and/or restoration of remnants with improved tree health. Stock and possum grazing pressure have been shown to influence tree health (Kirkpatrick *et al.* 2000) and eucalypt recruitment. Reduction of grazing pressure should lead to improved recruitment of eucalypts and other tree species. This could be promoted by the deliberate use of fire. In turn, this should lead to a vegetation structure more amenable to a diverse array of bird species, and these are likely to provide a more effective check on insect pests, thus reducing one source of stress on eucalypts and possibly improving overall tree health. This will not guarantee an improvement in tree health if other factors are culpable, and climate in particular is a thorny problem for an individual landowner to solve. However, the results of this study and others suggest that even if the primary cause of dieback is drought stress, that management practices as suggested above should ameliorate conditions for eucalypts. To some extent, the direct cause of dieback is not of paramount importance in terms of land management, as once eucalypts are weakened by one cause they are likely to be susceptible to others. Nevertheless, rehabilitation ought not to concentrate solely on establishment of eucalypts, which may result in a single-aged stand with little value for faunal elements. Wattles and she-oaks are extremely diverse groups and the different types of foliage, bark and seeds provided by these and other non-eucalypt trees provide extra foraging and food opportunities for birds (Recher 1985). Invertebrates may react to microclimate and soil differently from plants and conservation of vegetation will not automatically result in conservation of invertebrates (Greenslade & Greenslade 1984). However, results from the present study suggest that maintenance of forest remnants in relatively

undegraded condition should allow persistence of forest-dwelling grasshopper species. Species that prefer pastoral habitat are less likely to be at risk from the fragmentation and degradation of forests, although they may be at risk from other causes.

Most landowners and the public in general feel an affinity for the native fauna and are interested in its preservation. This study shows that to do so is not merely a case of leaving trees standing, that structure and condition of remnants is vital if they are not to become strongholds of noisy miners and half a dozen associated species. Forest patches on farmland may serve a number of purposes, and conservation of birds may conflict with some others, such as logging for firewood or stock shelter. Nevertheless, management of some remnants for bird conservation ought to be achievable and may have benefits for farmers in terms of pest control as well as for aesthetic reasons. Ultimately, remnants on private land form the greater part of the remaining woodland vegetation in Australia, and these remnants are under more intensive management than is most public land (Ford & Barrett 1995).

Maintenance of native vegetation has economic and aesthetic benefits for landowners. However, management of remnants for condition, in terms of vegetation, avifauna and invertebrate fauna may require a different regime from that which is now widespread. Grazing history has a significant impact on vegetation structure (Milchunas & Lauenroth 1993, Watson *et al.* 2000) and this in turn has been shown to affect the grasshopper and bird fauna of the Tasmanian Midlands in the present study. The value of possum-proofing trees and of fencing out stock for eucalypt recruitment and dieback mitigation has been shown in the Midlands (Kirkpatrick *et al.* 2000). The former at least is relatively inexpensive, while the latter will probably improve the vegetation structure of remnants in the long term, provided that conditions are suitable for regeneration, by establishing an understorey of non-eucalypt trees. Maintenance of an adequate amount of tree cover on the landscape scale may be more difficult, considering economic pressures on landowners. Bennett & Ford (1997) suggested 10% tree cover as a minimum goal to prevent serious decline in the avifauna of the Northern Plains in Victoria. Fensham & Kirkpatrick (1989) estimated that 83% of native vegetation in the northern Midlands had been converted to agricultural land in the Tasmanian Midlands, and further land clearance has occurred since then. While no bird species is known to have disappeared from the Midlands, the time lag associated with population decline

may mean that loss of species may occur in the future. Fortunately, revegetation programs by government and community associations such as Landcare are attempting to reverse the trend toward clearance of native vegetation.

The results of the present study combined with those elsewhere (Recher *et al.*1991) and those on the flora and vegetation of the study area (Kirkpatrick & Gilfedder 1995, Gilfedder & Kirkpatrick 1998) point strongly to the conclusion that conservation of habitat which is likely to conserve higher species richness of any group is of considerable benefit, but that the requirements of some individual species will not be met by such conservation methods. These species are likely to be those most sensitive to disturbances to natural habitat and will require more specific management. Future assessment of present conservation measures will not applaud the maintenance of high species richness; they will mourn the loss of elements of the fauna which occur currently.

9. References

ABBOTT, I. 1978: Density and species diversity of bird populations in *Eucalyptus* forests in Victoria, Bass Strait and Tasmania. *Proc. R. Soc. Vict.* **87**: 187-196.

ABENSPERG-TRAUN, M. 2000: In defence of small habitat islands: termites (Isoptera) in the Western Australian central wheatbelt, and the importance of dispersal power in species occurrence. *Pac. Conserv. Biol.* **6**: 31-39.

ABENSPERG-TRAUN, M., SMITH, G.T., ARNOLD, G.W. & STEVEN, D.E. 1996: The effects of habitat fragmentation and livestock-grazing on animal communities in remnants of gimlet *Eucalyptus salubris* woodland in the Western Australian wheatbelt. I Arthropods. *J. Appl. Ecol.* **33**: 1281-1301.

AMBUEL, B. & TEMPLE, S.A. 1983: Area-dependent changes in the bird communities and vegetation of southern Wisconsin forests. *Ecology* **64**: 1057-68.

ANDERSON, S.H., SHUGART, H.H. & SMITH, T.M. 1979: Vertical and temporal habitat utilization within a breeding bird community. pp in J.G. Dickson, R.N. Connor, R.R. Fleet, J.A. Jackson & J.C. Kroll (eds): *The role of insectivorous birds in forest ecosystems.* Academic Press, New York.

ANDERSON, R.V., TRACY, C.R. & ABRAMSKY, Z. 1979: Habitat selection in two species of short-horned grasshopper. *Oecologia* **38**: 359-374.

ANDREN, H. 1994: Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* **71**: 355-366.

ANDREN, H. & ANGELSTAM, P. 1988: Elevated predation rates as an edge effect in habitat islands: experimental evidence. *Ecology* **69**: 544-547.

ARNOLD, G.W., MALLER, R.A. & LITCHFIELD, R. 1987: Comparison of bird populations in remnants of wandoo woodland and in adjacent farmland. *Aust. Wildl. Res.* **14**: 331-341.

ARNOLD, G.W. & WEELDENBURG, J.R. 1998: The effects of isolation, habitat fragmentation and degradation by livestock grazing on the use by birds of patches of Gimlet *Eucalyptus salubris* woodland in the wheatbelt of Western Australia. *Pac. Conserv. Biol.* **4**: 155-163.

ARRHENIUS, O. 1921 Species and area. *J. Ecol.* **9**: 95-99.

ASKINS, R.A., PHILBRICK, M.J. & SUGENO, D.S. 1987: Relationship between the regional abundance of forest and the composition of forest bird communities. *Biol. Conserv.* **39**: 129-152.

BALDI, A. & KISBENEDEK, T. 1997: Orthopteran assemblages as indicators of grassland naturalness in Hungary. *Agric. Ecosyst. Environ.* **66**: 121-129.

BARRETT, G.F., FORD, H.A. & RECHER, H.F. 1994: Conservation of woodland birds in a fragmented rural landscape. *Pac. Conserv. Biol.* **1**: 245-256.

BELLAMY, P.E., HINSLEY, S.A.. & NEWTON, I. 1996: Factors influencing bird species numbers in small woods in south-east England. *J. Appl. Ecol.* **33**: 249-262.

BENNETT, A.F. & FORD, L.A. 1997: Land use, habitat change and the conservation of birds in fragmented rural environments: a landscape perspective from the Northern Plains, Victoria, Australia. *Pac. Conserv. Biol.* **3**: 244-261.

BENTLEY, J.M. & CATTERALL, C.P. 1997: The use of bushland, corridors and linear remnants by birds in southeastern Queensland, Australia. *Conserv. Biol.* **11**: 1173-1189.

BLAKE, J.G. & KARR, J.R. 1984: Species composition of bird communities and the conservation benefit of large versus small forests. *Biol. Conserv.* **30**: 173-187.

BLAKE, J.G. & KARR, J.R. 1987: Breeding birds of isolated woodlots: area and habitat relationships. *Ecology* **68**: 1727-1734.

BOECKLEN, W.J 1986: Effects of habitat heterogeneity on the species-area relationships of forest birds. *J. Biogeogr.* **13**: 59-68.

BOSWORTH 1976: Application of island biogeography principles to the selection and management of Tasmanian dry sclerophyll forest reserves. M. Env. St. Thesis, Uni of Tas, unpublished.

BOULINIER, T., NICHOLS, J.D., HINES, J.E., SAUER, J.R., FLATHER, C.H. & POLLOCK, K.H. 1998: Higher temporal variability of forest breeding bird communities in fragmented landscapes. *Proc. Natl. Acad. Sci. USA* **95:** 7497-7501.

BRAITHWAITE, L.W., AUSTIN, M.P., CLAYTON, M., TURNER, J. & NICHOLLS, A.O. 1989: On predicting the presence of birds in *Eucalyptus* forest types. *Biol. Conserv.* **50**: 33-50.

BRITTINGHAM, M.C. & TEMPLE, S.A. 1983: Have cowbirds caused forest songbirds to decline? *Bioscience* **33**: 31-35.

BUCKLEY, R. 1982: The habitat-unit model of island biogeography. *J. Biogeogr.* **9**: 339-344.

BURGESS, R.L. & SHARPE, D.M. 1981 (Eds.): *Forest island dynamics in man-dominated landscapes.* Springer-Verlag, New York.

CALE, P. 1994: Temporal changes in the foraging behaviour of insectivorous birds in a sclerophyll forest in Tasmania. *Emu* **94**: 116-126.

CAPINERA, J.L. & SECHRIST, T.S. 1982: Grasshopper (Acrididae) - host plant associations: response of grasshopper populations to cattle grazing intensity. *Can. Ent.* **114**: 1055-1062.

CATTERALL, C.P., KINGSTON, M.B. & PARK, K. 1997: Use of remnant forest habitat by birds during winter in subtropical Australia: patterns and processes. *Pac. Conserv. Biol.* **3**: 262-274.

CATTERALL, C.P., KINGSTON, M.B., PARK, K. & SEWELL, S. 1998: Deforestation, urbanisation and seasonality: interacting effects on a regional bird assemblage. *Biol. Conserv.* **84**: 65-81.

CIESLAK, M & DOMBROVSKI, A. 1993: The effect of forest size on breeding bird communities. *Acta Ornithol. Warsaw* **27**: 97-111.

CLARK, L.R. 1964: Predation by birds in relation to the population density of *Cardiaspina albitextura* (Psyllidae). *Aust. J. Zool.* **12**: 349-361.

CLARKE, M.F. 1984: Interspecific aggression within the genus *Manorina*. *Emu* **84**: 113-115.

CLARKE, M.F. 1988: The reproductive behaviour of the bell miner *Manorina melanophrys*. *Emu* **88**: 88-100.

CLARKE, M.F. & FITZGERALD, G.F. 1994: Spatial organization of the cooperatively breeding bell miner, *Manorina melanophrys*. *Emu* **94**: 96-105.

CLARKE, M.F. & SCHEDVIN, N. 1997: An experimental study of the translocation of noisy miners *Manorina melanocephala* and difficulties associated with dispersal. *Biol. Conserv.* **80**: 161-167.

CLARKE, M.F. & SCHEDVIN, N. 1999: Removal of bell miners *Manorina melanophrys* from *Eucalyptus radiata* forest and its effect on avian diversity, psyllids and tree health. *Biol. Conserv.* **88**: 111-120.

COLEMAN, B.D., MARES, M.A., WILLIG, M.R. & HSIEH, Y. 1982: Randomness, area, and species richness. *Ecology* **63**: 1121-1133.

CONNER, R.N. & DICKSON, J.G. 1980: Strip transect sampling and analysis for avian habitat studies. *Wildl. Soc. Bull.* **8**: 4-10.

CONNOR, E.F. & McCOY, E.D. 1979: The statistics and biology of the species-area relationship. *Am. Nat.* **113**: 791-833.

C.S.I.R.O. 1970: *The insects of Australia: a textbook for students and research workers.* Melbourne University Press.

DEACON, J.N. & Mac NALLY, R. 1998: Local extinction and nestedness of small-mammal faunas in fragmented forest of central Victoria, Australia. *Pac. Conserv. Biol.* **4**: 122-131.

den BOER, P.J. 1990: The survival value of dispersal in terrestrial arthropods. *Biol. Conserv.* **54**: 175-192.

DESROCHERS, A. & HANNON, S.J. 1997: Gap crossing decisions by forest songbirds during the post-fledging period. *Conserv. Biol.* **11**: 1204-1210.

DIAMOND, J.M., BISHOP, K.D. & van BALEN, S. 1987: Bird survival in an isolated Javan woodland: island or mirror? *Conserv. Biol.* **1**: 132-142.

DICKINSON, K.J.M., WALL, L.E. & WILSON, R.I. 1986: Birds in a partly clearfelled dry eucalypt forest on dolerite in southeastern Tasmania. *Pap. Proc. Roy. Soc. Tasm.* **120**: 39-49.

DIDHAM, R.K., GHAZOUL, J., STORK, N.E. & DAVIS, A.J. 1996: Insects in fragmented forests: a functional approach. *Trends in Evolution and Ecology* **11**: 255-260.

DONOVAN, T.M., JONES, P.W., ANNAND, E.M. & THOMPSON, F.R. (III) 1997: Variation in local-scale edge effects: mechanisms and landscape context. *Ecology* **78**: 2064-2075.

DOW, D.D. 1977: Indiscriminate interspecific aggression leading to almost sole occupancy of space by a single species of bird. *Emu* **77**: 115-121.

DOW, D.D. 1979: Agonistic and spacing behaviour of the noisy miner *Manorina melanocephala*, a communally breeding honeyeater. *Ibis* **121**: 423-436.

DUNCAN, F. 1999: Dry sclerophyll forests and woodlands. pp 244-264 in J.B. Reid, R.S. Hill, M.J. Brown & M.J. Hovenden (eds): *Vegetation of Tasmania.* Monotone Art Printers, Tasmania.

ER, K.B.H. 1997: Effects of eucalypt dieback on bird species diversity in remnants of native woodland. *Corella* **21**: 69-76.

EVANS, E.W. 1988: Grasshopper (Insecta: Orthoptera: Acrididae) assemblages of tallgrass prairie: influences of fire frequency, topography and vegetation. *Can. J. Zool.* **66**: 1495-1501.

FAHRIG, L. & MERRIAM, G. 1994: Conservation of fragmented populations. *Conserv. Biol.* **8**: 50-59.

FENSHAM, R.J. 1989: The pre-European vegetation of the Midlands, Tasmania: a floristic and historical analysis of vegetation patterns. *J. Biogeogr.* **16**: 29-45.

FENSHAM, R.J. & KIRKPATRICK, J.B. 1989: The conservation of original vegetation remnants in the Midlands, Tasmania. *Pap. Proc. Roy. Soc. Tasm.* **123**: 229-241.

FIELDING, D.J. & BRUSVEN, M.A. 1993: Spatial analysis of grasshopper density and ecological disturbance on southern Idaho rangeland. *Agric. Ecosyst. Environ.* **43**: 31-47.

FORD, H.A. 1981: Birds and eucalypt dieback. pp 112-120 in K.M. Old, G.A. Kile, & C.P. Ohmart (eds): *Eucalypt dieback in forests and woodlands.* CSIRO, Melbourne.

FORD, H.A. 1985: The bird community in eucalypt woodland and eucalypt dieback in the Northern Tablelands of New South Wales. pp 249-254. in J.A. Keast, H.F. Recher, H.A. Ford, & D. Saunders (eds): *Birds of eucalypt forests and woodlands: ecology, conservation, managements.* RAOU and Surrey Beatty, Chipping Norton.

FORD, H.A. 1986: Birds and eucalypt dieback in north-eastern New South Wales. pp in H.A. Ford & D.C. Paton (eds): *The dynamic partnership: birds and plants in southern Australia.* Government Printer, South Australia.

FORD, H.A. 1987: Bird communities on habitat islands in England. *Bird Study* **34**: 205-218.

FORD, H.A. 1989: *Ecology of birds: an Australian perspective.* Surrey Beatty and Sons, Chipping Norton.

FORD, H.A. & BARRETT, G. 1995: The role of birds and their conservation in agricultural systems. pp128-134 in A. Bennett, G. Backhouse & T. Clark (eds): *People and nature conservation. Perspectives on private land use and endangered species recovery.* Royal Zoological Society of New South Wales, Sydney.

FORD, H.A. & BELL, H.L. 1981: Density of birds in eucalypt woodland affected to varying degrees by dieback. *Emu* **81**: 202-208.

FORESTRY TASMANIA 2000: Plagued by wingless grasshoppers. *Forest Health Bulletin* **2**: 1-2.

FOX, L.R. & MORROW, P.A. 1983: Estimates of damage by herbivorous insects on *Eucalyptus* trees. *Aust. J. Ecol.* **8**: 139-147.

FREEMARK, K.E. & MERRIAM, G. 1986: Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. *Biol. Conserv.* **36**: 115-141

FRIESEN, L., CADMAN, M.D. & MacKAY, R.J. 1999: Nesting success of neotropical migrant songbirds in a highly fragmented landscape. *Conserv. Biol.* **13**: 338-346.

FRITH, H.J. 1977: *Waterfowl in Australia.* Angus & Robertson, Sydney.

GERBER, A.S. & TEMPLETON, A.R. 1996: Population sizes and within-deme movement of *Trimerotropis saxatilis* (Acrididae), a grasshopper with a fragmented distribution. *Oecologia* **105**: 343-350.

GIBSON, D.J. & HULBERT, L.C. 1987: Effects of fire, topography and year-to-year climatic variation on species composition in tallgrass prairie. *Vegetatio* **72**: 175-185.

GILFEDDER, L. & KIRKPATRICK J.B. 1998: Factors influencing the condition of natural vegetation remnants in subhumid Tasmania. *Biol. Conserv.* **84**: 89-96.

GLEASON, H.A. 1922: On the relation between species and area. *Ecology* **3**: 158-162.

GREEN, R.H. 1983: The decline of the eastern rosella and other Psittaciformes in Tasmania concomitant with the establishment of the introduced European starling. *Records of the Queen Victoria Museum* **82**: 1-5.

GREENSLADE, J. & GREENSLADE, P. 1984: Invertebrates and environmental assessment. *Environment and Planning* **3**:13-15.

GREY, M.J., CLARKE, M.F. & LOYN, R.H. 1997: Initial changes in the avian communities of remnant eucalypt woodlands following a reduction in the abundance of noisy miners, *Manorina melanocephala. Wildl. Res.* **24**: 631-648.

GREY, M.J., CLARKE, M.F. & LOYN, R.H. 1998: Influence of the noisy miner, *Manorina melanocephala,* on avian diversity and abundance in remnant grey box woodland. *Pacific Conserv. Biol.* **4**: 55-69.

HAGAN, J.M., VANDER HAEGEN, W.M. & McKINLEY, P.S. 1996: The early development of forest fragmentation effects on birds. *Conserv. Biol.* **10**: 188-202.

HAILA, Y. 1990: Toward an ecological definition of an island: a northwest European perspective. *J. Biogeogr.* **17**: 561-568.

HAILA, Y. & HANSKI, I.K. 1984: Methodology for studying the effect of habitat fragmentation on land birds. *Ann. Zool. Fenn.* **26**: 173-180.

HALME, E. & NIEMELA, J. 1993: Carabid beetles in fragments of coniferous forest. *Ann. Zool. Fenn.* **30**: 17-30.

HAMEL, P.B., SMITH, W.P. & WAHL, J.W. 1993: Wintering bird populations of fragmented forest habitat in the Central Basin, Tennessee. *Biol. Conserv.* **66**: 107-115.

HANSSON, L. 1997: Environmental determinants of plant and bird diversity in ancient oak-hazel woodland in Sweden. *For. Ecol. Manage.* **91**: 137-143.

HARDEN, R.H., MUIR, R.J. & MILLEDGE, D.R. 1986: An evaluation of the strip transect method for censusing bird communities in forests. *Aust. Wildl. Res.* **13**: 203-211.

HEATWOLE, H. & LOWMAN, M. 1986: *Dieback: death of an Australian landscape.* Reed Books, Frenchs Forest.

HELLE, P. & JARVINEN, O. 1986: Population trends of North Finnish land birds in relation to their habitat selection and changes in forest structure. *Oikos* **46**: 107-115.

HELLIWELL, D.R. 1976: The effects of size and isolation on the conservation value of woodland sites in Britain. *J. Biogeogr.* **3**: 407-416.

HIGGS, A.J. 1981: Island biogeographic theory and nature reserve design. *J. Biogeogr.* **8**: 117-124.

HOBBS, R.J. 1987: Disturbance regimes in remnants of natural vegetation. pp 233-240 in D.A. Saunders, G.W. Arnold, A.A. Burbidge & A.J.M. Hopkins (eds): *Nature*

conservation: the role of remnants of native vegetation. Surrey Beatty & Sons, Chipping Norton.

HOBBS, R.J. & HOPKINS, A.J.M 1990: From frontier to fragments: European impact on Australia's vegetation. *Proc. Ecol. Soc. Aust.* **16**: 93-114.

HOLMES, R.T., SCHULTZ, J.C. & NOTHNAGLE, P. 1979: Bird predation on forest insects: an exclosure experiment. *Science* **206**: 462-463.

HOWE, R.W. 1984: Local dynamics of bird assemblages in small forest habitat islands in Australia and North America. *Ecology* **65**: 1585-1601.

HOWE, R.W., DAVIS, G.J. & MOSCA, V. 1991: The demographic significance of "sink" populations. *Biol. Conserv.* **57**: 239-255.

HOWE, R.W., HOWE, T.D. & FORD, H.A. 1981: Bird distributions on small rainforest remnants in New South Wales. *Aust. Wildl. Res.* **8**: 637-651.

JANZEN, D.H. 1983: No park is an island: increase in interference from outside as park size decreases. *Oikos* **41**: 402-410.

KEAST, A. 1968: Seasonal movement in the Australian honeyeaters (Meliphagidae) and their ecological significance. *Emu* **67**: 159-210.

KEAST, A. 1985: Bird community structure in southern forests and northern woodlands: a comparison. pp 97-116 in J.A. Keast, H.F. Recher, H.A. Ford, & D. Saunders: *Birds of eucalypt forests and woodlands: ecology, conservation, managements.* RAOU and Surrey Beatty, Chipping Norton.

KENDEIGH, S.C. 1944: Measurement of bird populations. *Ecol. Monogr.* **14**: 68-106.

KILE, G.A. 1981: An overview of eucalypt dieback in rural Australia. pp 13-26 in K.M. Old, G.A. Kile, & C.P. Ohmart (eds): *Eucalypt dieback in forests and woodlands.* CSIRO, Melbourne.

KIRKPATRICK, J.B. & GILFEDDER, L. 1995: Maintaining integrity compared with maintaining rare and threatened taxa in remnant bushland in subhumid Tasmania. *Biol. Conserv.* **74**: 1-8.

KIRKPATRICK, J.B., GILFEDDER, L. & FENSHAM, R. 1988: *City parks and cemeteries.* Tasmanian Conservation Trust, Hobart.

KIRKPATRICK, J.B. & MARKS, F. 1985: Observation of drought damage to some native plant species in eucalypt forests and woodlands near Hobart, Tasmania. *Pap. Proc. Roy. Soc. Tasm.* **119**: 15-21.

KIRKPATRICK, J.B., ZACHAREK, A. & CHAPPELL, K. 2000: Testing methods for mitigation of tree dieback in Tasmanian dry eucalypt forests and woodlands. *Pac. Conserv. Biol.* **6**: 94-101.

KISBENEDEK, T. & Baldi, A. 1995: Island biogeography or orthopterans in steppe patches. Abstracts of the Seventh European Ecological Congress, p. 63.

KITCHENER, D.J. 1982: Predictors of vertebrate species richness in nature reserves in the Western Australian wheatbelt. *Aust. Wildl. Res.* **9**: 1-7.

KITCHENER, D.J., DELL, J., MUIR, B.G. & PALMER, M. 1982: Birds in Western Australian wheatbelt reserves - some implications for conservation. *Biol. Conserv.* **22**: 127-163.

KRUESS, A. TSCHARNTKE, T. 1994: Habitat fragmentation, species loss and biological control. *Science* **264**: 1581-1584.

LAMM, D.W. & CALABY, J.H. 1950: Seasonal variation of bird populations along the Murrumbidgee in the Australian Capital Territory. *Emu* **50**: 114-122.

LANDSBERG, J. 1988: Dieback of rural eucalypts: tree phenology and damage caused by leaf-feeding insects. *Aust. J. Ecol.* **13**: 251-267.

LANDSBERG, J., MORSE, J. & KHANNA, P. 1990: Tree dieback and insect dynamics in remnants of native woodland on farms. *Proc. Ecol. Soc. Aust.* **16**: 149-165.

LANDSBERG, J. & WYLIE, F.R. 1983: Water stress, leaf nutrients and defoliation: a model of dieback of rural eucalypts. *Aust. J. Ecol.* 8: 27-41.

LAURANCE, W.F. & YENSEN, E. 1991: Predicting the impacts of edge effects in fragmented habitats. *Biol. Conserv.* **55**: 77-92.

LEACH, G.J. 1996: Changes in populations of bird species in roadside softwood scrub remnants/farmland and open eucalypt forest in south-east Queensland, 1981 to 1993. *Pac. Conserv. Biol.* **2**: 232-243.

LEGGE, W.V. 1902: Note on the birds of Tasmania. *Pap. Proc. Roy Soc. Tasm. 1902*: 90-92.

LORD, J.M. & NORTON, D.A. 1990: Scale and the spatial concept of fragmentation. *Conserv. Biol.* **4**: 197-202.

LOW, T. 1994: Invasion of the savage honeyeaters. *Aust. Nat. Hist.* **24**: 26-33

LOWMAN, M.D. & HEATWOLE, H. 1992: Spatial and temporal variability in defoliation of Australian eucalypts. *Ecology* **73**: 129-142.

LOYN, R.H. 1985a: Ecology, distribution and density of birds in Victorian forests. pp 33-46 in J.A. Keast, H.F. Recher, H.A. Ford, & D. Saunders (eds): *Birds of eucalypt forests and woodlands: ecology, conservation, managements.* RAOU and Surrey Beatty, Chipping Norton.

LOYN, R.H. 1985b: Birds in fragmented forests in Gippsland, Victoria. pp 323-331 in J.A. Keast, H.F. Recher, H.A. Ford, & D. Saunders (eds): *Birds of eucalypt forests and*

woodlands: ecology, conservation, managements. RAOU and Surrey Beatty, Chipping Norton.

LOYN, R.H. 1987: Effects of patch area and habitat on bird abundances, species numbers and tree health in fragmented Victorian forests. pp 65-77 in D.A. Saunders, G.W. Arnold, A.A. Burbidge & A.J.M. Hopkins (eds): *Nature conservation: the role of remnants of native vegetation.* Surrey Beatty & Sons, Chipping Norton.

LOYN, R.H. 1993: Effects of previous logging on bird populations in East Gippsland. *VSP Technical Report No. 18.*

LOYN, R.H. & MIDDLETON, W.G.D. 1981: Eucalypt decline and wildlife in rural areas. pp95-111 in K.M. Old, G.A. Kile, & C.P. Ohmart (eds): *Eucalypt dieback in forests and woodlands.* CSIRO, Melbourne.

LOYN, R.H., RUNNALLS, R.G., FORWARD, G.Y. & TYERS, J. 1983: Territorial bell miners and other birds affecting populations of insect prey. *Science* **221**: 1411-1413.

LUCK, G.W., POSSINGHAM, H.P. & PATON, D.C. 1999a: Bird responses at inherent and induced edges in the Murray mallee, South Australia. 1. Differences in abundance and diversity. *Emu* **99**: 157-169.

LUCK, G.W., POSSINGHAM, H.P. & PATON, D.C. 1999a: Bird responses at inherent and induced edges in the Murray mallee, South Australia. 2. Nest predation as an edge effect. *Emu* **99**: 170-175.

LYNCH, J.L. 1987: Responses of breeding bird communities to forest fragmentation. pp 123-140 in D.A. Saunders, G.W. Arnold, A.A. Burbidge & A.J.M. Hopkins (eds). *Nature conservation: the role of remnants of native vegetation.* Surrey Beatty & Sons, Chipping Norton.

LYNCH, J.P. & WHIGHAM, D.F 1984: Effects of forest fragmentation on breeding bird communities in Maryland, USA. *Biol. Conserv.* **28**: 287-324.

MacARTHUR, R.H., MacARTHUR, J.W. & PREER, J. 1962: On bird species diversity II. Prediction of bird census from habitat analysis. *Am. Nat.* **96**: 167-174.

MacARTHUR, R.H. & WILSON, E.O. 1967: *The theory of island biogeography.* Princeton University Press, Princeton.

Mac NALLY, R.C. 1990b: The roles of floristics and physiognomy in avian community composition. *Aust. J. Ecol.* **15**: 321-327.

Mac NALLY, R.C. 1994: Habitat-specific guild structure of forest birds in south-eastern Australia: a regional scale perspective. *J. Anim. Ecol.* **63**: 988-1001.

MAGURRAN, A.E. 1988: *Ecological diversity and its measurement.* University Press, Cambridge.

MAIN, B.Y. 1987: Persistence of invertebrates in small areas: case studies of trapdoor spiders in Western Australia. pp 29-39 in D.A. Saunders, G.W. Arnold, A.A. Burbidge & A.J.M. Hopkins (eds). *Nature conservation: the role of remnants of native vegetation.* Surrey Beatty & Sons, Chipping Norton.

MAJER, J.D., RECHER, H. & KEALS, N. 2000: Canopy arthropod faunas in fragmented agricultural landscapes. pp 235-247 in R. Hobbs & C. Yates (eds). *Temperate eucalypt woodlands in Australia.* Surrey Beatty & Sons, Chipping Norton.

MARGULES, C.R., MILKOVITS, G.A.. & SMITH, G.T. 1994: Contrasting effects of habitat fragmentation on the scorpion *Cercophonius squama* and an amphipod. *Ecology* **75**: 2033-2042.

MARK, D. & STUTCHBURY, B.J. 1994: Response of a forest-interior songbird to the threat of cowbird parasitism. *Anim. Behav.* **47**: 275-280.

MAYFIELD, H. 1977: Brown-headed cowbird: agent of extermination? *Am. Birds* **31**: 107-113.

McCOLLIN, D. 1993: Avian distribution patterns in a fragmented wooded landscape (North Humberside, U.K.): the role of between-patch and within-patch structure. *Global Ecology and Biogeography Letters* **3**: 48-62.

McCOY, E.D. 1982: The application of island-biogeographic theory to forest tracts: problems in the determination of turnover rates. *Biol. Conserv.* **22**: 217-227.

McINTYRE, S. & BARRETT, G.W. 1992: Habitat variegation, and alternative to fragmentation. *Conserv. Biol.* **6**: 146-147.

McQUILLAN, P. 1982: Wingless grasshopper. *Farmnotes* **118**. p.1.

MILCHUNAS, D.G. & LAUENROTH, W.K. 1993: Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecol. Monogr.* **63**: 327-366.

MINCHIN, P.R. 1990: *DECODA - Database for Ecological Community Data, version 2.02.* Australian National University.

MINITAB, INC. 1997a: User's guide 1: data graphics and macros.

MINITAB, INC. 1997b: User's guide 2: data analysis and quality tools.

NEW, T.R. 1992: *Entomology for Australian students.* New South Wales University Press.

NEWMAN, O.M.G. 1983: The RAOU field atlas asa data source for Tasmanian birds. *Tasmanian Bird Report* **12**: 4-8.

NILSSON, S.G. 1986: Are bird communities in small biotope patches random samples from communities in large patches? *Biol. Conserv.* **38**: 179-204.

OLD, K.M., KILE, G.A. & OHMART, C.P. (Eds) 1981: *Eucalypt dieback in forests and woodlands.* CSIRO, Melbourne.

OPDAM, P., RIJSDIJK, G. & HUSTINGS, F. 1985: Bird communities in small woods in an agricultural landscape: effects of area and isolation. *Biol. Conserv.* **34**: 333-352.

OPDAM, P., van DORP, D. & ter BRAAK, C.J.F. 1984: The effect of isolation on the number of woodland birds in small woods in the Netherlands. *J. Biogeogr.* **11**: 473-476.

ORANGE-BELLIED PARROT RECOVERY TEAM 1999: Orange-bellied parrot recovery plant 1998-2002. Department of Primary Industry, Water and Environment, Tasmania.

OTTE, D. 1976: Species richness patterns of New World desert grasshoppers in relation to plant diversity. *J. Biogeogr.* **3**: 197-209.

OTVOS, I.S. 1979: The effects of insectivorous bird activities in forest ecosystems: an evaluation. pp 341-374. in J.G. Dickson, R.N. Connor, R.R. Fleet, J.A. Jackson & J.C. Kroll (eds): *The role of insectivorous birds in forest ecosystems.* Academic Press, New York.

PANZER, R., STILLWAUGH, D., GNAEDINGER, R. & DERKOVITZ, G. 1995: Prevalence of remnant dependence among the prairie- and savanna-inhabiting insects of the Chicago region. *Nat. Areas J.* **15**: 101-116.

PATON, D.C. 1980: The importance of manna, honeydew and lerp in the diet of honeyeaters. *Emu* **80**: 213-226.

PATON, P.W. 1994: The effect of edge on avian nest success: how strong is the evidence? *Conserv. Biol.* **8**: 17-26.

PETTIT, N.E., FROEND, R.H. & LADD, P.G. 1995: Grazing in remnant woodland vegetation: changes in species composition and life form groups. *J. Veg. Sci.* **6**: 121-130.

PODGER, F.D. 1981: Definition and diagnosis of diebacks. pp 1-9 in K.M. Old, G.A. Kile, & C.P. Ohmart (eds): *Eucalypt dieback in forests and woodlands.* CSIRO, Melbourne.

POIANI, A. 1993: Bell miners: what kind of 'farmers' are they? *Emu* **93**: 188-194.

POIANI, A. 1995: On bell miners, farmers and the role of metaphors in science: a reply to Loyn. *Emu* **95**: 147-148.

POIANI, A., ROGERS, A., ROGERS, K. & ROGERS, D. 1990: Asymmetrical competition between the bell miner (*Manorina melanophrys*, Meliphagidae) and other honeyeaters: evidence from southeastern Australia. *Oecologia* **85**: 250-256.

POLDMAA, T., MONTGOMERIE, R. & BOAG, P. 1995: Mating system of the cooperatively breeding noisy miner *Manorina melanocephala*, as revealed by DNA profiling. *Behav. Ecol. Sociobiol.* **37**: 137-143.

QUINN, J.F. & HASTINGS, A. 1987: Extinction in subdivided habitats. *Conserv. Biol.* **1**: 198-208.

QUINN, M.A., KEPNER, R.L., WALGENBACH, D.D., BOHLS, R.A., POOLER, P.D., FOSTER, R.N., REUTER, K.C. & SWAIN, J.L. 1991: Habitat characteristics and grasshopper community dynamics on mixed-grass rangeland. *Can. Entomol.* **123**: 89-105.

QUINN, M.A. & WALGENBACH, D.D. 1990: Influence of grazing history on the community structure of grasshoppers of a mixed-grass prairie. *Environ. Entomol.* **19**: 1756-1766.

RAFE, R.W., USHER, M.B. & JEFFERSON, R.G. 1985: Birds on reserves: the influence of area and habitat on species richness. *J. Appl. Ecol.* **22**: 327-335.

RATKOWSKY, A. 1983: Birds of the Mount Wellington bush. *Tasmanian Bird Report* **12**: 4-8.

RATKOWSKY, A.V. & RATKOWSKY, D.A. 1979: A comparison of counting methods to obtain bird species numbers. *Notornis* **26**: 53-61.

RECHER, H.F. 1985: Eucalypt forests, woodlands and birds: an introduction. pp 1-10 in J.A. Keast, H.F. Recher, H.A. Ford, & D. Saunders (eds): *Birds of eucalypt forests and woodlands: ecology, conservation, managements.* RAOU and Surrey Beatty, Chipping Norton.

RECHER, H.F., KAVANAGH, R.P., SHIELDS, J.M. & LIND, P. 1991: Ecological associations of habitats and bird species during the breeding season in southeastern New South Wales. *Aust. J. Ecol.* **16**: 337-352.

RENTZ, D.C.F. : *Grasshopper country: the abundant orthopteroid insects of Australia.* University of New South Wales Press, Sydney.

RIDPATH, M.G. & MOREAU, R.E. 1966: The birds of Tasmania: ecology and evolution. *Ibis* **108**: 348-393.

ROBBINS, C.S. 1980: Effect of forest fragmentation on breeding bird populations in the piedmont of the mid-Atlantic region. *Atl. Nat.* **33**: 31-36.

ROBBINS, C.S., DAWSON, D.K. & DOWELL, B.A. 1989: Habitat area requirements of breeding forest birds of the Middle Atlantic States. *Wildl. Monogr.* **103**: 1-34.

ROBINSON, G.R., HOLT, R.D., GAINES, M.S., HAMBURG, S.P., JOHNSON, M.L., FITCH, H.S. & MARTINKO, E.A. 1992: Diverse and contrasting effects of habitat fragmentation. *Science* **257**: 524-526.

ROBINSON, S.K., THOMPSON, F.R. III, DONOVAN, T.M., WHITEHEAD, D.R. & FAABORG, J. 1995: Regional forest fragmentation and the nesting success of migratory birds. *Science* **267**: 1987-1990.

ROLSTAD, J. 1991: Consequences of forest fragmentation for the dynamics of bird populations: conceptual issues and the evidence. *Bio. J. Linn. Soc.* **42**: 149-163.

ROWLEY, I. 1965: The life history of the superb blue wren, *Malurus cyaneus. Emu* **64**; 251-297.

SAUNDERS, D.A. & CURRY, P.J. 1990: The impact of agricultural and pastoral industries on birds in the southern half of Western Australia: past, present and future. *Proc. Ecol. Soc. Aust.* **8**: 303-321.

SAUNDERS, D.A., HOBBS, R.J. & MARGULES, C.R. 1991: Biological consequences of ecosystem fragmentation: a review. *Conserv. Biol.* **5**: 18-32.

SCHIECK, J., LERTZMA, K., NYBERG, B., & PAGE, R. 1995: Effects of patch size on birds in old-growth montane forests. *Conserv. Biol.* **9**: 1072-1084.

SCOUGALL, S.A., MAJER, J.D. & HOBBS, R.J. 1993: Edge effects in grazed and ungrazed Western Australian wheatbelt remnants in relation to ecosystem reconstruction. pp 163-178 in D.A. Saunders, R.J. Hobbs & P.R. Ehrlich (eds). *Nature conservation 3: reconstruction of fragmented ecosystems, global and regional perspectives.* Surrey Beatty & Sons, Chipping Norton.

SEMMENS, T.D., McQUILLAN, P.B., & HAYHURST, G. (1992): Catalogue of the insects of Tasmania. Dept. Primary Industry, Hobart. 104 pp.

SEWELL, S.R. & CATTERALL, C.P. 1998: Bushland modification and styles of urban development: their effects on birds in south-east Queensland. *Wildl. Res.* **25**: 41-63.

SIMBERLOFF, D. & ABELE, L.G. 1982: Refuge design and island biogeographic theory: effects of fragmentation. *Am. Nat.* **112**: 713-726.

SIMBERLOFF, D.S., FARR, J.A., COX, J. & MEHLMAN, D.W. 1992: Movement corridors: conservation bargains or poor investments? *Conserv. Biol.* **6**: 493-504.

SISK, T.D. & MARGULES, C.R. 1993: Habitat edges and restoration: methods for quantifying edge effects and predicting the results of restoration efforts. pp 57-69 in D.A. Saunders, R.J. Hobbs & P.R. Ehrlich (eds): *Nature conservation 3: reconstruction of fragmented ecosystems.* Surrey Beatty & Sons, Chipping Norton

SLATER, P.J. 1995: The interaction of bird communities with vegetation and season in Brisbane Forest Park. *Emu* **95**: 194-207.

SMITH, A.J. & ROBERTSON, B.I. 1978: Social organization of bell miners. *Emu* **78**: 169-178.

STACEY, P.B. & TAPER, M. 1992: Environmental variation and the persistence of small populations. *Ecol. Appl.* **2**: 18-29.

STONE, C. 1996: The role of psyllids (Hemiptera: Psyllidae) and bell miners (*Manorina melanophrys*) in canopy dieback of Sydney blue gum (*Eucalyptus saligna* Sm.). *Aust. J. Ecol.* **21**: 450-458.

SZARO, R.C. 1986: Guild management: an evaluation of avian guilds as a predictive tool. *Environ. Manage.* **10**: 681-688.

TAYLOR, B.L. 1995: The reliability of using population viability analysis for risk classification of species. *Conserv. Biol.* **9**: 551-558.

TAYLOR, R., DUCKWORTH, P., JOHNS, T. & WARREN, B. 1997: Succession in bird assemblages over a seven-year period in regrowth dry sclerophyll forest in south-east Tasmania. *Emu* **97**: 220-230.

TELLERIA, J.L. & SANTOS, T. 1995: Effects of forest fragmentation on a guild of wintering passerines: the role of habitat selection. *Biol. Conserv.* **71**: 61-67.

TEMPLE, S.A. & CAREY, J.R. 1988: Modelling dynamics of habitat interior bird populations in fragmented landscapes. *Conserv. Biol.* **2**: 340-347.

TEWKSBURY, J.J., HEJL, S.J. & MARTIN, T.E. 1998: Breeding productivity does not decline with increasing fragmentation in a western landscape. *Ecology* **79**: 2890-2903.

THOMAS, D.G. 1974: Some problems associated with the avifauna. pp 339-365 in W.D. Williams (ed) *Biogeography and ecology in Tasmania.* Dr W. Junk, The Hague.

THOMAS, D.G. 1979: *Tasmanian bird atlas.* University of Tasmania, Hobart.

THOMAS, D.G. 1980: Foraging of honeyeaters in an area of Tasmanian sclerophyll forest. *Emu* **80**: 55-58.

THOMAS, D.G. 1986: The birds of Mount Wellington - comparison of two 10 ha plots of dry and wet sclerophyll. *Tasmanian Bird Report* **15**: 11-16.

van DORP, D. & OPDAM, P. 1987: Effects of patch size, isolation and regional abundance on forest bird communities. *Landscape Ecol.* **4**: 171-176.

VILLARD, M., MERRIAM, G. & MAURER, B.A. 1995: Dynamics in subdivided populations of neotropical migratory birds in a fragmented temperate forest. *Ecology* **76**: 26-40.

WATTS, D. 1999: *Field guide to Tasmanian birds.* New Holland Publishers, Sydney.

WATSON, D.M., Mac NALLY, R. & BENNETT, A.F. 2000: The avifauna of severely fragmented buloke *Allocasuarina luehmanni* woodland in western Victoria, Australia. *Pac. Conserv. Biol.* **6**: 46-60.

WEBB, N.R. 1989: Studies on the invertebrate fauna of fragmented heathland in Dorset, UK, and the implications for conservation. *Biol. Conserv.* **47**: 153-165.

WEDDELL, B.J. 1991: Distribution and movements of Columbian ground squirrels (*Spermophilus columbianus* (Ord)): are habitat patches like islands? *J. Biogeogr.* **18**: 385-394.

WHITCOMB, R.F., ROBBINS, C.S., LYNCH, J.F., WHITCOMB, B.L., KLIMKIEWICZ, M.K. & BYSTRAK, D. 1981: Effects of forest fragmentation on avifauna of the eastern deciduous forest. pp 125-206 in R.L. Burgess & D.M. Sharpe (eds): *Forest island dynamics in man-dominated landscapes.* Springer-Verlag, New York.

WIENS, J.A. 1989: *The ecology of bird communities.* Cambridge University Press, Cambridge.

WILCOX, B.A., MURPHY, D.D., EHRLICH, P.R. & AUSTIN, G.T. 1986: Insular biogeography of the montane butterfly faunas in the Great Basins: comparison with birds and mammals. *Oecologia* **69**: 188-194.

WILLIAMS, C.B. 1964: *Patterns in the balance of nature.* Academic Press, London.

WILLIS, E.O. 1984: Conservation, subdivision of reserves, and the anti-dismemberment hypothesis. *Oikos* **42**: 396-398.

WOINARSKI, J.C.Z. 1984: Small birds, lerp-feeding and the problem of honeyeaters. *Emu* **84**: 137-141.

WOINARSKI, J.C.Z. 1985: Foliage-gleaners of the treetops, the pardalotes. pp 165-175 in J.A. Keast, H.F. Recher, H.A. Ford, & D. Saunders (eds): *Birds of eucalypt forests and woodlands: ecology, conservation, managements.* RAOU and Surrey Beatty, Chipping Norton

WYLIE, F.R. & LANDSBERG, J. 1990: Rural dieback. pp 243-249 in Cremer, K.W. (ed): *Trees for rural Australia.* Inkatha Press, Melbourne.

YATES, C.J. & HOBBS, R.J. 1997: Temperate eucalypt woodlands: a review of their status, processes threatening their persistence and techniques for restoration. *Aust. J. Bot.* **45**: 949-973.

Appendix One: Study sites

Remnant Name	Grid Reference	1:25000 Map	Size (ha)	Dominant eucalypt	Bird spp.	G'hopper spp.
Punchbowl	5138 54107	Launceston	34.00	*E. viminalis*	31	3
Rufus Lagoon	5038 53975	Longford	129.50	*E. amygdalina*	29	5
Woodstock Lagoon	5042 53965	Longford	64.75	*E. amygdalina*	33	5
Kelton	5080 53911	Longford	13.25	*E. pauciflora*	19	1
Powranna	5216 53845	Nile	5.00	*E. amygdalina*	19	3
Fairfield Cressy	5158 53799	Delmont	13.25	*E. viminalis*	19	8
Pisa	5103 53728	Delmont	11.75	*E. amygdalina*	21	4
Carnavon	5135 53713	Delmont	21.75	*E. amygdalina*	15	3
Barton West	5178 53672	O'Connors	15.25	*E. amygdalina*	15	3
Barton South	5182 53667	O'Connors	16.75	*E. amygdalina*	14	4
Broadwater	5195 53718	Delmont	21.00	*E. viminalis*	24	4
Valleyfield	5257 53737	Cleveland	6.50	*E. amygdalina*	15	5
Fairfield Epping	5268 53777	Cleveland	5.00	*E. pauciflora*	15	3
Forton West	5242 53813	Nile	197.00	*E. amygdalina*	35	2
Forton Middle	5252 53804	Nile	7.75	*E. amygdalina*	11	1
Forton East	5257 53802	Nile	32.50	*E. amygdalina*	30	4
Esk Vale	5282 53782	Cleveland	79.00	*E. amygdalina*	31	4
York Park	5368 53796	Cleveland	81.75	*E. amygdalina*	32	4
Middle Run Hill	5360 53727	Cleveland	21.25	*E. viminalis*	27	*
Vaucluse West	5371 53714	Cleveland	5.25	*E. amygdalina*	11	2
Vaucluse East	5380 53714	Cleveland	4.00	*E. amygdalina*	7	1
Greenhill	5248 53644	Conara	26.75	*E. pauciflora*	26	7
Quorn Hall	5469 53538	C. Town	43.00	*E. viminalis*	16	6
Chiswick	5450 53495	Ross	21.50	*E. pauciflora*	14	6
Frankston	5293 53464	Ellinthorp	6.75	*E. viminalis*	11	7
Annandale North	5285 53387	Tunbridge	39.00	*E. amygdalina*	13	2
Annandale South	5284 53378	Tunbridge	95.00	*E. amygdalina*	36	2
Bald Hill	5268 53322	Tunbridge	24.50	*E. amygdalina*	29	2
Braeside West	5331 53157	Oatlands	9.50	*E. viminalis*	8	3
Braeside East	5337 53143	Oatlands	50.25	*E. viminalis*	9	6
Dull Hill	5333 53132	Oatlands	104.75	*E. viminalis*	30	5
Lowick	5395 53111	Oatlands	41.50	*E. viminalis*	14	2
Mother Lords Hill	5109 52963	Kempton	59.75	*E. viminalis*	28	4
Tarella	5097 52908	Kempton	21.25	*E. amygdalina*	32	4

Montacute Hill	4905	52975	Montacute	42.75	*E. tenuiramis*	25	7
Rathlyn	4937	52865	Hamilton	12.75	*E. tenuiramis*	12	4
Steels Hill	4931	52812	Hamilton	81.00	*E. tenuiramis*	29	5
Norton Mandeville	4915	52798	Bushy Park	79.50	*E. tenuiramis*	33	5
Brooksby	5212	52748	Teatree	39.00	*E. viminalis*	31	4
Glen Quoin	5244	52699	Richmond	46.25	*E. amygdalina*	14	5
Poimena	5197	52635	New Norfolk	35.25	*E. viminalis*	30	3
Queen's Domain	5263	52545	Hobart	104.00	*E. viminalis*	22	*
Boronia Hill	5254	52402	Taroona	45.25	*E. amygdalina*	25	7
Big Bush	5325	52646	Richmond	20.50	*E. viminalis*	12	1
Gordons Hill	5295	52545	Hobart	47.25	*E. viminalis*	26	1
Barilla	5402	52582	Carlton	5.50	*E. viminalis*	16	2
Clifton	5440	52419	Cremorne	11.50	*E. amygdalina*	13	3
E.amygdalina Centre	5245	53752	Cleveland	2808.50	*E. amygdalina*	27	5
E.amygdalina Edge	5267	53752	Cleveland	2808.50	*E. amygdalina*	25	3
E.pauciflora Centre	5249	53747	Cleveland	2808.50	*E. pauciflora*	19	2
E.tenuiramis Centre	5044	53064	Bothwell	590.50	*E. tenuiramis*	26	5
E.tenuiramis Edge	5034	53063	Bothwell	590.50	*E. tenuiramis*	10	3
E.viminalis Centre	5266	52632	Richmond	6849.75	*E. viminalis*	25	*

Appendix Two: Bird species recorded

Nomenclature follows Watts (1999).

Family	Scientific Name	Common Name (* denotes exotic species)	Migratory Guild[a]	Foraging Guild[b]	Food Preference Guild[c]	Fragment-ation Response[d]	No. of sites	Total number observed
Ardeidae	*Egretta novaehollandiae*	white-faced heron	R	*	*	*	5	12
Anatidae	*Tadorna tadornoides*	Australian shelduck	N	*	*	*	6	17
Anatidae	*Anas castanea*	chestnut teal	N	*	*	*	3	4
Anatidae	*Chenonetta jubata*	Australian wood duck	N	*	*	*	6	57
Accipitridae	*Accipiter fasciatus*	brown goshawk	R	R	V, I	3	16[e]	23[e]
Accipitridae	*Accipiter cirrhocephalus*	collared sparrowhawk	R	R	V, I	3	16[e]	23[e]
Accipitridae	*Aquila audax*	wedge-tailed eagle	R	R	V	*	6	6
Accipitridae	*Circus approximans*	swamp harrier	M	R	V	*	2	2
Falconidae	*Falco peregrinus*	peregrine falcon	R	R	V	*	3	3
Falconidae	*Falco berigora*	brown falcon	R	R	V, I	1	18	37
Falconidae	*Falco longipennis*	Australian hobby	R	R	V, I	*	1	1
Falconidae	*Falco cenchroides*	nankeen kestrel	R	R	V, I	*	1	1
Rallidae	*Gallinula mortierii*	Tasmanian native-hen	R	*	*	*	1	3
Charadriidae	*Vanellus miles*	masked lapwing	R	GC	I	*	3	23

Family	Scientific Name	Common Name (* denotes exotic species)	Migratory Guild[a]	Foraging Guild[b]	Food Preference Guild[c]	Fragment-ation Response[d]	No. of sites	Total number observed
Columbidae	*Phaps chalcoptera*	common bronzewing	R	G	S	1	31	193
Cacatuidae	*Calyptorhynchos funereus*	yellow-tailed black cockatoo	N	BP	I	*	3	7
Cacatuidae	*Cacatua galerita*	sulphur-crested cockatoo	R	G	S	1	28	667
Cacatuidae	*Cacatua roseicapilla*	galah	N	G	S	*	3	7
Loriidae	*Glossopsitta concinna*	musk lorikeet	N	N	N, I	5	15	100
Platycercidae	*Lathamus discolor*	swift parrot	M	N	N, I	*	3	49
Platycercidae	*Neophema chrysostoma*	blue-winged parrot	N	G	S	*	5	20
Platycercidae	*Platycercus caledonicus*	green rosella	R	G	S	2	43	753
Platycercidae	*Platycercus eximius*	eastern rosella	R	G	S	1a	41	1792
Cuculidae	*Cuculus pallidus*	pallid cuckoo	M	P	I	4	19	38
Cuculidae	*Cacomantis flabelliformis*	fan-tailed cuckoo	M	P	I	3	11	30
Cuculidae	*Chrysococcyx basalis*	Horsfield's bronze-cuckoo	M	F	I	*	1	2
Cuculidae	*Chrysococcyx lucidus*	shining bronze-cuckoo	M	F	I	3	12	31
Alcedinidae	*Dacelo novaeguineae*	laughing kookaburra*	R	P	V, I	1	44	360
Hirundidae	*Hirundo neoxena*	welcome swallow	M	I	S	1	14	22
Hirundidae	*Hirundo nigricans*	tree martin	M	I	S	*	2	16
Motacillidae	*Anthus novaseelandiae*	Richard's pipit	N	GC	I, S	*	1	2
Campephigidae	*Coracina novaehollandiae*	black-faced cuckoo-shrike	M	W	I	4	36	218
Muscicapidae	*Turdus merula*	common blackbird*	R	GC	I	5	15	246

Family	Scientific Name	Common Name (* denotes exotic species)	Migratory Guild[a]	Foraging Guild[b]	Food Preference Guild[c]	Fragment-ation Response[d]	No. of sites	Total number observed
Muscicapidae	*Petroica phoenicea*	flame robin	N	P	I	3	11	30
Muscicapidae	*Petroica multicolor*	scarlet robin	R	P	I	4	27	354
Muscicapidae	*Melanodryas vittata*	dusky robin	N	P	I	4	9	58
Muscicapidae	*Pachycephala pectoralis*	golden whistler	R	W	I	4	22	114
Muscicapidae	*Colluricincla harmonica*	grey shrike-thrush	R	BP	I	4	24	306
Muscicapidae	*Rhipidura fuliginosa*	grey fantail	N	H	I	3	31	1415
Maluridae	*Malurus cyaneus*	superb fairy-wren	R	GC	I	3	37	1646
Acanthizidae	*Acanthiza pusilla*	brown thornbill	R	F	I	3	33	3263
Acanthizidae	*Acanthiza chrysorrhoa*	yellow-rumped thornbill	N	GC	I	3	31	1094
Meliphagidae	*Anthochaera paradoxa*	yellow wattlebird	N	N	I	4	25	402
Meliphagidae	*Anthochaera chrysoptera*	brush wattlebird	R	N	I, N	*	5	7
Meliphagidae	*Manorina melanocephala*	noisy miner	R	W	I	1a	47	4147
Meliphagidae	*Lichenostomus flavicollis*	yellow-throated honeyeater	R	F	I	4	28	1088
Meliphagidae	*Melithreptus affinis*	black-headed honeyeater	N	F	I	*	6	210
Meliphagidae	*Phylidonyris pyrrhoptera*	crescent honeyeater	N	W	I, N	4	23	460
Meliphagidae	*Phylidonyris novaehollandiae*	New Holland honeyeater	R	N	I, N	*	2	8
Meliphagidae	*Acanthorhynchus tenuirostris*	eastern spinebill	N	N	I, N	*	6	21

Family	Scientific Name	Common Name (* denotes exotic species)	Migratory Guild[a]	Foraging Guild[b]	Food Preference Guild[c]	Fragment-ation Response[d]	No. of sites	Total number observed
Pardalotidae	*Pardalotus punctatus*	spotted pardalote	N	F	I	3	36	846
Pardalotidae	*Pardalotus striatus*	striated pardalote	M	F	I	1	49	2860
Zosteropidae	*Zosterops lateralis*	silvereye	M	BC	I	3	22	387
Fringillidae	*Carduelis carduelis*	European goldfinch*	N	G	S	5	16	169
Passeridae	*Passer domesticus*	house sparrow*	R	G	I, S	*	3	48
Sturnidae	*Sturnus vulgaris*	common starling*	R	GC	I	1	53	4170
Artamidae	*Atramus cyanopterus*	dusky woodswallow	M	H	I	3	21	300
Cractidae	*Cracticus torquatus*	grey butcherbird	R	P	V, I	1a	45	386
Cractidae	*Gymnorhina tibicen*	Australian magpie	R	GC	I	1a	43	913
Cractidae	*Strepera fuliginosa*	black currawong	N	GC	O	*	2	5
Cractidae	*Strepera versicolor*	grey currawong	N	BP	O	2	29	103
Corvidae	*Corvus tasmanicus*	forest raven	N	GC	O	1	52	1028

[a] R = resident, M = summer migrant, N = nomad

[b] *sensu* Mac Nally (1994): S = sweeper, H = hawker, P = pouncer, GC = ground carnivore, BC = bush carnivore, BP = bark prober, W = wood searcher, F = foliage searcher, N = nectarivore, G = granivore. R = raptors, * = waterbird not included in analysis.

[c] I = invertebrates, V = vertebrates, S = seeds, N = nectar, O = omnivorous, * = waterbird not included in analysis.

[d] 1 = miner-tolerant, unaffected by fragmentation (1a = strongly associated with miner colonies), 2 = miner-tolerant, susceptible to fragmentation, 3 = miner-intolerant, unaffected by fragmentation, 4 = miner-intolerant, susceptible to fragmentation, 5 = miner-intolerant, advantaged by fragmentation.

[e] Note that the brown goshawk and collared sparrowhawk were considered one species for this study. Combined values are shown here.

Appendix Three: Grasshopper species recorded

Nomenclature follows Semmens *et al.* (1992).

Family	Scientific Name	No. of sites	Total number collected
Acrididae	*Austroicetes frater*	12	24
Acrididae	*Austroicetes pusilla*	5	8
Acrididae	*Austroicetes vulgaris*	30	106
Acrididae	*Cirphula pyrrhocnemis*	2	3
Acrididae	*Exarna includens*	3	7
Acrididae	*Gastrimargus musicus*	2	6
Acrididae	*Goniaea australasiae*	11	18
Acrididae	*Macrotona australis*	18	45
Acrididae	*Oedalius australis*	12	39
Acrididae	*Peakesia brunniana*	2	2
Acrididae	*Phaulacridium nanum*	3	9
Acrididae	*Phaulacridium vittatum*	47	930
Acrididae	*Russalpia albertisi*	6	18
Acrididae	*Tasmaniacris tasmaniensis*	31	223
Acrididae	*Urnisa rugosa*	6	20